FINAL
REMEDIAL PROCESS OPTIMIZATION
PHASE II EVALUATION REPORT
FOR DEFENSE DEPOT MEMPHIS, TENNESSEE

Prepared for:

US AIR FORCE CENTER FOR ENVIRONMENTAL EXCELLENCE
CONSULTANT OPERATIONS DIVISION (AFCEE/ERC)
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and

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ENVIRONMENTAL AND SAFETY OFFICE (DSS-E)
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EXECUTIVE SUMMARY

Parsons Engineering Science, Inc. (Parsons ES) was contracted by the Defense Logistics Agency Environmental Safety and Policy Office (DLA/CAAE) and the Air Force Center for Environmental Excellence Consultant Operations Division (AFCEE/ERC) to conduct a remedial process optimization (RPO) evaluation for Defense Depot Memphis, Tennessee (DDMT). The general goals for each site addressed under DLA's RPO program are to: 1) assess the effectiveness of selected remedies; 2) enhance the effectiveness and efficiency of the remedies; and 3) when possible, identify optimization opportunities that could result in annual operating, maintenance, and/or monitoring cost savings for the systems evaluated.

DDMT was placed on the National Priorities List in October 1992. The Base Realignment and Closure (BRAC) Commission announced in July 1995 that the Depot had been selected for closure in September 1997. As part of the BRAC process, the buildings, facilities, and land at the Depot are being evaluated for transfer to the City of Memphis for subsequent commercial uses.

The RPO process is focused on two study areas at the Depot: the Main Installation (MI) and Dunn Field. The Draft Record of Decision (ROD) for the MI was issued in January 2001 (CH2M Hill, 2001a). The nature, extent, and magnitude of contamination at Dunn Field has not yet been fully characterized, and feasibility studies (FSs) have not been prepared; however, an interim ROD for restoration of groundwater quality at Dunn Field was approved in April 1996, and a groundwater recovery system at the downgradient boundary of this site, including 11 groundwater extraction wells, is currently operating.

Most soil contamination at the MI has been or will soon be remediated. Groundwater contamination beneath both the MI and Dunn Field consists primarily of chlorinated volatile organic compounds (CVOCs). The highest CVOC concentrations associated with the MI groundwater plumes have been detected upgradient from (southwest of) the MI, and appear to be migrating toward the northeast beneath the MI; the source(s) of this contamination have not been determined. Dissolved CVOC contamination in Dunn Field groundwater appears to be primarily related to former waste disposal pits. CVOCs dissolved in groundwater have migrated off the installation to the west. Elevated CVOC concentrations have been detected in vadose zone soils at Dunn Field, and are the likely source of the groundwater contamination.

The following tasks were completed in conjunction with the RPO evaluation for DDMT:

- Assist in the development of an optimized soil vapor extraction (SVE) pilot test design for Dunn Field vadose zone soils using all available site characterization data;

- Review the natural attenuation indicator parameter data collected and reported by CH2M Hill (2000f) and evaluate the conclusions of the natural attenuation evaluation;

- Develop decision trees to establish clear operational and closure strategies for remedial action and groundwater monitoring activities;

- Review existing geochemical data for groundwater, and the results of vegetable oil injection pilot tests at other installations, to assess the advantages and disadvantages of using this technology at DDMT;

- Use appropriate models and statistical packages to evaluate the stability of the CVOC plumes in groundwater originating near the southwestern corner of the MI;

- Review the existing groundwater monitoring plan for Dunn Field, describe a statistical methodology that could be used for optimization of the monitoring program, and implement the methodology as appropriate;

- Evaluate and provide recommendations regarding the use of passive diffusion-bag (PDB) groundwater samplers at the MI and Dunn Field to support future long-term monitoring (LTM) of CVOCs;

- Develop schedule-to-complete (STC) and cost-to-complete (CTC) estimates to aid in future planning;

- Review the current regulatory environment under which remedial actions at the site are being performed, and assess the appropriateness of applicable or relevant and appropriate requirements, remedial action objectives, or remedial goals at the site; and

- Prepare an RPO Phase II evaluation report presenting the results of the above-described tasks and recommendations for optimizing future remediation efforts at DDMT.

- The results of these tasks are summarized below.

SVE PILOT TEST RECOMMENDATIONS

The shallow loess layer beneath the Depot has a relatively low permeability; this type of soil is often considered to be a poor candidate for SVE without some form of soil fracturing to improve the secondary porosity and open channels for air-flow. SVE pilot testing is required to evaluate the effectiveness of SVE in the loess. The sand that extends from the base of the loess to the water table is very well suited for SVE.

Two nested vapor extraction wells (VWs) were installed at Dunn Field in October 2000 near a previously-identified contamination “hot spot”. The shallow and deep wells are screened entirely within the loess and fluvial sand formations, respectively. These two VWs were constructed to be the test wells during a pilot test, and also could be used as part of a full-scale SVE system. Four multi-depth vapor monitoring points (VMPs) were also installed at varying distances from the VWs. Each VMP consists of four screened intervals within the same borehole separated by bentonite seals.

SVE pilot tests should be performed separately for the two VWs, but the two tests can be conducted during the same mobilization. During each pilot test, subsurface pressures and soil gas parameters should be monitored at the four screened intervals of each of the VMPs and at a background location. Monitoring the pilot test blower system, subsurface pressure distribution, and soil gas parameters will aid in determining SVE system design parameters for the loess and the fluvial sand. Installation of two additional VMPs, and temporary conversion of an existing groundwater monitoring well to a VMP, is recommended prior to pilot testing. SVE pilot testing should be performed ASAP to support the remedial decision-making process and accelerate attainment of cleanup goals for soil and groundwater.

DECISION TREE DEVELOPMENT

Decision trees were developed for: 1) groundwater monitoring program optimization; 2) groundwater extraction system operation and shutdown; and 3) SVE system operation and shutdown. The objective of the decision trees is to aid in future decision-making, while moving toward the ultimate goal of site cleanup.

The groundwater monitoring decision tree outlines how to assess whether or not an existing well, or a new well being considered for installation, should be included in a monitoring network by completing a review of site information and performing temporal and spatial analyses using qualitative and/or statistical techniques. Guidance on selecting appropriate sampling frequencies also is provided.

The groundwater extraction decision tree provides the user with guidance regarding (1) the activities that should be performed to support the effectiveness of the remedial system; (2) determining when remedial goals have been met or if they can they be met, and (3) determining when the system (or a portion of the system) can be shut down. Similarly, the SVE decision tree assists the user to identify the steps needed to optimize a SVE system in order to reach soil cleanup goals, and to determine when a system can be shut down.

REVIEW OF NATURAL ATTENUATION INDICATOR PARAMETER DATA

In general, CH2M Hill's (2000f) conclusion that only limited and localized biologically-facilitated reductive dehalogenation of CVOCs is occurring at the MI and Dunn Field appears to be reasonable and correct. The Dunn Field site exhibits a slightly greater potential for biodegradation of CAHs than the MI based on the results of the scoring process described by the US Environmental Protection Agency (USEPA) (1998) and Wiedemeier *et al*. (1999). However, the data supporting the occurrence of biodegradation do not outweigh the data that indicate that the CAH plumes at both sites can be classified as Type III, meaning that the groundwater system is characterized by inadequate concentrations of native and/or anthropogenic carbon, and concentrations of dissolved oxygen are greater than 1 milligram per liter. These conditions are not supportive of reductive dehalogenation of tetrachloroethene (PCE) and trichloroethene (TCE).

Based on first-order biodegradation rates computed using the method of Buscheck and Alcantar (1995), the maximum PCE concentration detected on the MI in March 2000 (78 micrograms per liter [μg/L]) would decrease to the maximum contaminant level (MCL) of 5 μg/L in 34 to 113 years (years 2035 and 2114, respectively). The discrepancy between these remediation time frames is caused by the use of differing fraction organic carbon (f_{oc}) concentrations in the decay rate calculations; this discrepancy highlights the utility and necessity of clarifying the f_{oc} content of the Fluvial aquifer. The estimated time frame for the maximum TCE concentration detected in the Dunn Field plume in March 2000 (1,200 μg/L) to decrease to the MCL of 5 μg/L under the influence of natural attenuation is 38 years (year 2039). The remedial time frame estimates for TCE and PCE assume that the contaminant source has been effectively removed, which is not the case for Dunn Field, and may not be true for the MI. Therefore, reliance on MNA alone would potentially require a commitment to many years of groundwater monitoring if groundwater cleanup goals remain equivalent to drinking water standards.

ASSESSMENT OF ENHANCED BIOREMEDIATION

The use of enhanced bioremediation could prove to be a cost-effective engineered remedial measure for MI groundwater. If it can be concluded from a pilot study that enhanced bioremediation is effective at the MI, then this technology could be used to reduce CVOC concentrations within the plume, and if the source is identified and removed, then enhanced bioremediation potentially could be used to remediate groundwater to cleanup goals. The time required to reach the PCE cleanup goal at the MI is reduced to 13 to 17 years based on estimation of the degree to which PCE decay rates would be enhanced (and assuming that the remaining source is insignificant or removed).

Vegetable oil injection is an innovative, cost-effective method of carbon addition that promotes the oxidation/reduction and electron-donor conditions necessary to promote *in situ* microbial dehalogenation of solvents in groundwater. Vegetable oil has several potential advantages compared to the Regenesis product HRC®, including lower upfront cost and lower solubility and mobility. Therefore, performance of a vegetable oil injection pilot test, either instead of or in addition to an HRC® pilot test, is recommended. Vegetable oil injection has been performed at several sites across the US including Naval Support Activity Mid-South in Memphis, Tennessee. At the Mid-South facility, the results of the first round of post-injection sampling conducted 3 months after injection indicate that the groundwater system in the immediate vicinity of the injection wells is becoming more reducing, methane is being produced, and reductive dehalogenation is being stimulated.

Source location activities should be implemented ASAP using the SimulProbe™ or similar device to support optimal location of enhanced bioremediation pilot test wells and the long-term effectiveness and permanence of the selected remedy. The enhanced bioremediation pilot test should be performed as soon as sufficient information is available to select optimal pilot test well locations.

EVALUATION OF MAIN INSTALLATION PLUME STABILITY

BIOCHLOR modeling was performed to assess whether the PCE and TCE plumes near the southwestern corner of the MI could potentially migrate to monitoring well MW34 at concentrations of concern. Previous studies have indicated the potential presence of a "window" or gap in the clay layer underlying the Fluvial aquifer in the vicinity of MW34 that could allow migration of groundwater and dissolved contaminants into deeper zones. Given the alternate conceptual groundwater flow model for the Fluvial aquifer presented in the MI ROD (CH2M Hill, 2001a), a secondary modeling objective was to assess the potential for chlorinated solvents dissolved in groundwater near the southwestern corner of the MI to migrate in a southeasterly direction toward installation boundary well MW24.

In addition to the BIOCHLOR modeling, statistical tests were performed to assess the stability of the chlorinated solvent plumes in the southwestern corner of the MI. The statistical algorithms used to accomplish this objective included the MAROS software package (AFCEE, 2000) and an alternate algorithm developed by Parsons ES.

Based on the results of the BIOCHLOR modeling and the statistical analyses, further expansion of the PCE and TCE plumes in the southwestern corner of the MI cannot be ruled out. However, plume expansion, if it occurs, should not pose a significant risk to off-site receptors (i.e., the Allen Well Field or potential receptors south of well MW24).

REVIEW OF DUNN FIELD GROUNDWATER MONITORING PROGRAM

The current groundwater monitoring program for Dunn Field was evaluated to identify potential opportunities to streamline monitoring activities while maintaining an effective program that monitors the performance of the groundwater extraction system and the potential for contaminants to migrate beyond the system. The approach used in this evaluation is presented on the groundwater monitoring decision tree described above. This approach involves evaluating the importance of each well in the monitoring network and its sampling frequency by using a combination of qualitative and statistical (temporal and spatial) analyses.

Based on the results of this evaluation, the majority of the 20 wells being monitored at Dunn Field are appropriate for inclusion in the monitoring network. Only 4 of the 20 wells should be considered for exclusion from the monitoring program, either because they appear to be redundant or because they are not particularly useful for evaluating the plume over time. However, this conclusion is based only on monitoring results for TCE. Therefore, the conclusions derived from the evaluation are preliminary, and are presented as an example of the recommended evaluation approach. Other COCs should be included in this evaluation in the future for a more complete evaluation of the monitoring program.

Reduction of the sampling frequency to semi-annual or annual is recommended, depending on the well location. Based on the recent monitoring data for Dunn Field obtained by Parsons ES, organochlorine pesticides and semivolatile organic compounds have generally not been detected in groundwater samples, and deletion of these analytes from the target list should be considered unless sampling of new wells in less-characterized areas is initiated. These recommendations should be considered ASAP in light of the most recent groundwater quality data obtained, and implemented as appropriate.

DIFFUSION SAMPLING EVALUATION

PDB samplers for VOC monitoring utilize passive sampling techniques that eliminate the need for well purging. These samplers are typically water-filled containers that are initially deployed within a screened interval of a well. Over an equilibration period (typically at least two weeks), the concentration of VOCs within the PDB sampler reaches equilibrium with VOC concentrations in the surrounding groundwater due to diffusion across a semi-permeable membrane. Following the equilibration period, the PDB sampler is retrieved from the well, and the water from within the diffusion sampler is transferred to a conventional sample container and submitted to a laboratory for analysis.

Based on estimated costs for PDB and conventional sampling at Dunn Field, a cost savings of approximately $160 per well per sampling event could be realized using PDB samplers. Evaluation of field testing of PDB samplers is recommended for LTM of VOCs at DDMT based on the demonstrated effectiveness at other sites, and on the potential cost savings over conventional purge sampling methods. The initial PDB sampling event should include vertical profiling and a thorough evaluation of the comparability of PDB sampling results with results from the current purge sampling methods. This evaluation would consist of performing a side-by-side comparison of PDB sampling results with conventional sampling results collected during the same monitoring event. This initial verification would result in additional monitoring expenses for this first event (estimated to be approximately $40,000 for a work plan, vertical profiling of 20 wells at Dunn Field, and a results report comparing PDB sampling results with conventional results).

The recommended vertical profiling and comparison with conventional sampling should be performed ASAP in the portion of Dunn Field that is currently monitored on a regular basis if the planned sampling frequencies indicate that significant cost-savings would be achieved. This recommendation should be considered for implementation at the MI once the scope of the final LTM programs (e.g., numbers and locations of wells, analyte list, and sampling frequencies) are determined. In this way, a more accurate assessment of potential cost savings and the desirability of PDB sampling can be determined.

EVALUATION OF REMEDIATION GOALS

The selected remedy for soils at the MI will likely achieve acceptable risk levels and allow the property to be transferred or leased for its intended land use. The achievement of drinking water standards in groundwater at the MI is less certain based on the potential limitations of existing remedial technologies and the lack of source identification. If the source(s) of the CVOC plumes beneath the southwestern portion of the MI cannot be found, and if future groundwater monitoring indicates that dissolved CVOC concentrations both on and upgradient from the MI are not decreasing at an acceptable rate, then classification of MI groundwater as Site Specific Impaired should be sought. If granted, alternate, risk-based cleanup goals should be developed. This recommendation should be considered as the 5-year review of the MI ROD approaches. Available data should be reviewed 1 year prior to the 5-year review to allow assessment of progress toward source characterization and achievement of groundwater cleanup goals.

A recommended procedure for using a 1-D analytical model to derive soil cleanup goals that are protective of groundwater quality is presented. The purpose of site-specific, unsaturated-zone contaminant transport modeling for Dunn Field would be to evaluate the possible downward migration of chemicals of concern (COCs) through the vadose zone to the water table, and to predict the maximum concentrations of these COCs that could remain in the vadose zone without allowing their continued migration to the water table at concentrations that would exceed groundwater cleanup levels (e.g., MCLs). The results of this evaluation can then be used to calculate the concentrations of COCs in the vapor phase, in equilibrium with the maximum sorbed and dissolved soil concentrations, that could remain in the soil column within the vadose zone. These calculated vapor-phase concentrations of COCs would represent screening-level indicators of site-specific cleanup criteria for COCs in soil at Dunn Field that could be used to assess compliance with soil cleanup goals using soil gas data collected from the SVE system. Measured VOC concentrations in soil gas, obtained during SVE pilot testing activities, should be compared to recommended cleanup criteria in the SVE pilot test results report.

COST-TO-COMPLETE AND SCHEDULE-TO-COMPLETE

The estimated cost to complete remediation for DDMT is approximately $20.1 million. This includes estimated costs for fiscal year 2001 and beyond for anticipated completion of remediation activities. The CTC is summarized on a total cost basis (i.e., not discounted for present worth).

The schedule-to-complete (STC) for the MI extends to 2019. The STC for Dunn Field extends to 2040. Calculated cleanup time frames are sensitive to site-specific input parameters that may not be well-defined, so the actual STC may vary significantly from these predictions. In addition, the STC for Dunn Field will depend on the final remedy selected for groundwater.

SUMMARY

Table ES.1 provides a summary of the optimization recommendations, and potential cost savings associated with their implementation, as identified during the RPO evaluation for DDMT. Assumptions undergirding the cost saving estimates are provided in the table footnotes. If all recommendations were implemented, an estimated total cost savings of $1.6 million could be realized. Additional, though unquantified, savings could accrue from location and remediation of the contaminant source(s) near the southwestern portion of the MI (Recommendation No. 7). Implementation suggestions for the RPO opportunities are included in Section 12 of this document.

TABLE ES.1
REMEDIAL OPTIMIZATION RECOMMENDATIONS SUMMARY
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Recommendation	Estimated Annual Cost Savings[a]	Cost Savings Over Life Cycle[a]	Difficulty of Implementation	Estimated Cost to Implement[a]
Recommendation 1: Perform a SVE pilot test at Dunn Field prior to completion of FS reports.	NA[b]	$225 K[c/ d]	Low	$5 K[e]
Recommendation 2: Develop vadose zone soil cleanup levels for Dunn Field that are based on the relationship between average equilibrium soil gas concentrations in the soil column and the groundwater protection or risk-based soil standard for the site.	NA	$15 K[f]	Moderate – Requires regulatory approval	$10 K
Recommendation 3: Perform vegetable oil pilot test and evaluate use of this approach for groundwater remediation at the MI.	NA	$664 K[g]	Low to Moderate – Based on approval of this approach at Navy Mid-South Site in Millington	$6 K[h]
Recommendation 4: Consider deletion of four monitoring wells from current Dunn Field monitoring program and adjust sampling frequencies to semiannual or annual.	$1.6 K[i]	$47 K[i]	Moderate – Requires regulatory approval.	$5 K
Recommendation 4: Consider deletion of SVOCs and pesticides from target analyte list.	$8.4 K[j]	$252 K[j]	Moderate – Requires regulatory approval	$1 K
Recommendation 5: Clarify fraction organic carbon content of the Fluvial aquifer to better predict cleanup time frames and plume migration distances.	NA	NA	Low	$1 K
Recommendation 6: Evaluate use of PDB samplers at Dunn Field, and phase in their use at the MI in the future as appropriate.	$9.6 K[k]	$216K[k]	Moderate – Requires regulatory approval.	$50 K
Recommendation 7: Perform contaminant source location activities near SW corner of MI to facilitate cleanup of dissolved plumes.[l]	TBD[m]	TBD	Moderate – Requires significant field investigation	$200 K
Recommendation 8: Seek classification of MI groundwater as Site Specific Impaired and develop alternate, risk-based cleanup levels if future monitoring indicates that drinking water standards are not obtainable.	$25 K[n]	$201 K[n]	Moderate – Requires regulatory approval	$15 K

TABLE ES.1
REMEDIAL OPTIMIZATION RECOMMENDATIONS SUMMARY
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Recommendation	Estimated Annual Cost Savings[a/]	Cost Savings Over Life Cycle[a/]	Difficulty of Implementation	Estimated Cost to Implement[a/]
TOTAL	$44.6 K	$1.62 M		$288 K

[a/] Estimated costs given in constant 2001 dollars.
[b/] NA = not applicable.
[c/] Assume remediation timeframe for Dunn Field is shortened by 1 year, resulting in savings of $59,050 for long-term monitoring and $167,236 for remedial system O&M. (see Table 11.1).
[d/] K = Thousand dollars. M = Million dollars.
[e/] Cost assumes that a SVE pilot test will eventually be performed; therefore, the pilot test cost does not represent an extra, unscoped cost. Cost shown is for regulatory interaction.
[f/] Assumes that drilling footage for confirmation soil sampling is reduced by 400 feet @ $30/foot due to the soil gas evidence that soil cleanup criteria have been met.
[g/] Assumes that cost savings are derived from difference in materials cost between HRC® and vegetable oil of $5.65/lb and from lower annual costs in years 2-5 as a result of only performing a single injection of vegetable oil as opposed to annual injections of HRC®. Cost savings for pilot test is (900 lb x $5.65/lb) = $5,085. Cost savings for year 1 of full-scale remediation is (10,400 lb x $5.65/lb) = $58,760. Cost savings for years 2-5 of full-scale remediation (based on Table 2-14 in MI ROD) is (4 x $58,760 for materials) + (4 x $63,000 for installation) + (4 x $38,400 for labor) + (4 x $1,000 for mob/demob) + (4 x $4,800 for pump) = $663,840. Cost savings could be higher if remediation time frame exceeds 5 years.
[h/] Cost assumes that an enhanced bioremediation pilot test will be performed; therefore, the pilot test cost does not represent an extra, unscoped cost. Cost shown is for development of a detailed vegetable oil pilot test cost estimate and regulatory interaction.
[g/] Assumes that cost savings are derived solely from difference in materials cost between HRC and vegetable oil of $5.65/lb. Assumes that a single injection of vegetable oil is sufficient for 5 years of treatment. Cost savings for pilot test is 900 lb x $5.65/lb = $5,085. Cost savings for year 1 of full-scale remediation is 10,400 lb x $5.65/lb = $58,760. Cost savings for years 2-5 of full-scale remediation is 4 x $58,760 = $235,040. Cost savings will be higher if remediation time frame exceeds 5 years.
[i/] Assumes that deleted wells are sampled annually for VOCs, SVOCs, and pesticides for 30 years. Cost savings include labor ($720 per event), laboratory analyses ($485 per event), drums ($50 per event), and data validation/reporting ($300 per event).
[j/] Assumes deletion of SVOCs ($225 per sample) and pesticides ($125 per sample) for 20 wells sampled annually for 30 years. Assumes 4 QC samples per event.
[k/] Estimated savings per well per event = $160 (Table 9.1). Assumes 20 wells sampled for 30 years at Dunn Field , half annually and half semi-annually. Assumes 20 wells sampled for 15 years at MI, half annually and half semi-annually.
[l/] The Tennessee Department of Environment and Conservation has stated that they are performing a site investigation to look for potential sources of upgradient, offsite groundwater contaminants.
[m/] TBD = To be determined.
[n/] Assumes that Site Specific Impaired Status obtained in 2010, and long-term monitoring costs from 2011-2018 (Table 11.1) are reduced by 67 percent as a result.

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LIST OF ACRONYMS AND ABBREVIATIONS

α_x, α_y, α_z	Dispersivity in three dimensions
ρ_b	Bulk density
1-D	One-dimensional
3-D	Three-dimensional
λ	First-order decay rate
µg/L	Micrograms per liter
ACC	Air Combat Command
AFBCA	Air Force Base Conversion Agency
AFCEE/ERC	Air Force Center for Environmental Excellence, Consultant Operations Division
ARAR	Applicable or Relevant and Appropriate Requirement
ASTM	American Society for Testing and Materials
BCT	BRAC Cleanup Team
bgs	Below ground surface
BOD	Biological oxygen demand
BRA	Baseline Risk Assessment
BRAC	Base Realignment and Closure
CAAE	DLA Environmental and Safety Policy Office
CAIS	Chemical Agent Identification Sets
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
cm/sec	Centimeters per second
COC	Chemical of concern
CO_2	Carbon dioxide
CPT	Cone penetrometer testing
CPPT	Critical path planning toolbox
CSM	Conceptual site model
CTC	Cost-to-complete
CVOC	Chlorinated volatile organic compound
CWM	Chemical warfare materiel
DCA	Dichloroethane
DCE	Dichloroethene
DDC	Defense Distribution Center
DDHU	Defense Depot Hill, Utah
DDMT	Defense Depot Memphis, Tennessee
DLA	Defense Logistics Agency
DNAPL	Dense nonaqueous-phase liquid
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DOD	Department of Defense
DRC	Depot redevelopment corporation
EE/CA	Engineering Evaluation/Cost Analysis
ESE	Environmental Science and Engineering, Inc.
f_{oc}	Fraction of organic carbon
FRTR	Federal Remediation Technologies Roundtable
FS	Feasibility Study

ft/day	Feet per day
ft/ft	Foot per foot
ft/yr	Feet per year
FU	Functional unit
FY	Fiscal year
GAC	Granular activated carbon
GIS	Geographic Information System
HELP	Hydrologic evaluation of landfill performance
HRC	Hydrogen-releasing compound
I	hydraulic gradient
IA	Installation Assessment
IDW	Investigation-derived waste
IRP	Installation Restoration Program
K	hydraulic conductivity
K_{oc}	Soil adsorption coefficient
L/kg	Liter per kilogram
LTM	Long-term monitoring
MAROS	Monitoring and Remediation Optimization System
MCL	Maximum contaminant level
mg/kg	Milligrams per kilogram
mg/L	Milligrams per liter
MI	Main installation
MIP	Membrane Interface Probe™
ml	milliliter
MNA	Monitored natural attenuation
n_e	Effective porosity
NE	Northeast
NPL	National Priorities List
O_2	Oxygen
OM&M	Operation, maintenance, and monitoring
O&M	Operation and maintenance
ORP	Oxidation-reduction potential
OU	Operable unit
OVA	Organic vapor analyzer
OVA/FID	Organic vapor analyzer/flame ionization detector
PAHs	Polynuclear aromatic hydrocarbons
Parsons ES	Parsons Engineering Science, Inc.
PCA	Tetrachloroethane
PCBs	Polychlorinated biphenyls
PCE	Perchloroethene (tetrachloroethene)
PCP	Pentachlorophenol
PDB	Passive diffusion bag
POC	Point of Compliance
ppmv	Parts per million by volume
PVC	Polyvinyl chloride
R	Retardation coefficient
RAB	Restoration Advisory Board
RAO	Remedial action objective

RBCA	Risk based corrective action
Redox	Reduction/oxidation
RG	Remedial Goals
RI	Remedial investigation
RNA	Remediation by natural attenuation
ROD	Record of Decision
RPO	Remedial Process Optimization
RSV	RPO Scoping Visit
scfm	standard cubic feet per minute
SVE	soil vapor extraction
STC	schedule-to-complete
SVOCs	Semivolatile organic compounds
$t_{1/2}$	half-life
TCA	Trichloroethane
TCE	Trichloroethene
TCDD	Tetrachlorodibenzo-p-dioxin
TDEC	Tennessee Department of Environment and Conservation
TI	Technical impracticibility
TRW	Technical review workgroup
USACE	US Army Corps of Engineers
USAEHA	US Army Environmental Hygiene Agency
USAESCH	US Army Engineering & Support Center, Huntsville
USATHAMA	United States Army Toxic and Hazardous Materials Agency
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
VC	Vinyl chloride
VMP	Vapor monitoring point
VOA	Volatile organics analysis
VOCs	Volatile organic compounds
VW	Vapor extraction well
V	Advective groundwater velocity

SECTION 1

INTRODUCTION

Parsons Engineering Science, Inc. (Parsons ES) was awarded task order TG03 under Air Combat Command (ACC) contract F44650-99-D0005 on 21 March 2000, to support remedial process optimization (RPO) scoping visits (RSVs) and to conduct RPO Phase II evaluations at selected Defense Logistics Agency (DLA) facilities. The Defense Depot Memphis, Tennessee (DDMT), one of the facilities included in the contract, is the subject of this RPO Phase II evaluation report. The Headquarters DLA Environmental and Safety Policy Office (CAAE) has initiated the RPO program to evaluate existing and planned environmental remediation systems with the intention of identifying and implementing changes to improve the efficiency and effectiveness of those systems. The US Air Force Center for Environmental Excellence, Consultant Operations Division (AFCEE/ERC) is providing technical oversight for the task order.

The RPO approach is described as a three-phase process in the draft *Air Force Remedial Process Optimization Handbook* (AFCEE and Air Force Base Conversion Agency [AFBCA], 1999). Phase I consists of an annual review of site cleanup objectives, remedial system performance, and progress toward achieving cleanup goals. Phase II is an intensive evaluation to explore system optimization, new technologies, and/or regulatory opportunities. For sites such as DDMT, at which environmental restoration is governed by a record of decision (ROD), Phase II evaluations should occur at least 1 year prior to mandatory 5-year ROD reviews. Phase III consists of implementing the opportunities developed during Phase I and/or Phase II evaluations. The benefits of RPO can include reduced operation, maintenance, and monitoring (OM&M) costs while maintaining adequate protection of human health and the environment; reevaluation of cleanup goals; better tracking of remediation progress; and accelerated site closure.

The RSV is an additional tool for identifying RPO opportunities. The purpose of an RSV is to evaluate the overall effectiveness of remediation systems and monitoring programs, identify sites and/or remedial systems that could benefit from Phase II evaluations, and make specific recommendations for areas to be addressed during follow-up RPO assessments. The specific objectives of this RPO Phase II evaluation were developed during the RSV conducted at the Depot in June 2000. The RSV team included representatives from DLA/CAAE, AFCEE/ERC, the Defense Distribution Center (DDC), Mitretek Systems, Booz•Allen & Hamilton, and Parsons ES. During the RSV, the team interacted extensively with personnel from the DDMT, CH2M Hill, and the US Army Corps of Engineers Engineering and Support Center-Huntsville (USAESCH). The potential RPO opportunities identified during the RSV were discussed at the project kickoff meeting, which was held in Denver, Colorado on 11 July 2000, and the scope of the Phase II evaluation was solidified.

1.1 FACILITY-SPECIFIC SCOPE AND OBJECTIVES

DDMT was placed on the National Priorities List (NPL) in October 1992. The Base Realignment and Closure (BRAC) Commission announced in July 1995 that the Depot had been selected for closure in September 1997. As part of the BRAC process, the buildings, facilities, and land at the Depot are being evaluated for transfer to the City of Memphis for subsequent commercial uses.

The RPO process is focused on two study areas at the Depot: the Main Installation (MI) and Dunn Field. The Draft ROD for the MI was issued in January 2001 (CH2M Hill, 2001a). The nature, extent, and magnitude of contamination at Dunn Field has not yet been fully characterized, and feasibility studies (FSs) have not been prepared; however, an interim ROD for restoration of groundwater quality at Dunn Field was approved in April 1996, and a groundwater recovery system, including 11 groundwater extraction wells, is currently operating at the site.

The RPO Phase II work plan for DDMT (Parsons ES, 2000) outlined the objectives and activities to be completed during the RPO Phase II evaluation for the Depot. Based on the information available for consideration during this project, and the operational status of the remedial systems at the facility, the final objectives of the RPO Phase II evaluation for this Depot were as follow:

- Assist in the development of an optimized soil vapor extraction (SVE) pilot test design for Dunn Field vadose zone soils using all available site characterization data;
- Review the natural attenuation indicator parameter data collected and reported by CH2M Hill (2000c) and evaluate the conclusions of the natural attenuation evaluation;
- Develop decision trees to establish clear operational and closure strategies for all remedial action and groundwater monitoring activities;
- Review existing geochemical data for groundwater, and the results of vegetable oil injection pilot tests at other installations, to assess the advantages and disadvantages of using this technology at DDMT;
- Use appropriate models and statistical packages to evaluate the stability of the plume of chlorinated volatile organic compound (CVOC) contamination in groundwater originating near the southwestern corner of the MI;
- Review the existing groundwater monitoring plan for Dunn Field, describe a statistical methodology that could be used for optimization of the monitoring program, and implement the methodology as appropriate;
- Evaluate and provide recommendations regarding the use of passive diffusion-bag (PDB) groundwater samplers at the MI and Dunn Field to support future long-term monitoring (LTM) of CVOCs;

- Develop schedule-to-complete (STC) and cost-to-complete (CTC) estimates to aid in future planning;

- Review the current regulatory environment under which remedial actions at the site are being performed, and assess the appropriateness of applicable or relevant and appropriate requirements (ARARs), remedial action objectives (RAOs), or remedial goals (RGs) at the site; and

- Prepare an RPO Phase II evaluation report presenting the results of the above-described tasks and recommendations for optimizing future remediation efforts at DDMT.

The results of each of these tasks, along with the specific activities performed to complete these tasks, are described in detail in subsequent sections of this report.

1.2 REPORT ORGANIZATION

This RPO Phase II evaluation report is organized into 13 sections, including this introduction, and 5 appendices. A review of the facility history and other background information is presented in Section 2. Section 3 describes the results of soil gas sampling and SVE well installation at Dunn Field, and presents recommendations for conducting an SVE pilot test. Section 4 presents decision trees for groundwater monitoring program optimization, and the operation and closure of groundwater extraction and SVE systems, together with accompanying descriptive narratives. A review of the natural attenuation evaluation performed by CH2M Hill (2000c) is contained in Section 5, and Section 6 contains an assessment of enhanced bioremediation via vegetable oil injection. The stability of the chlorinated solvent plume originating near the southwestern corner of the MI is evaluated in Section 7 using the results of BIOCHLOR modeling and statistical tests. The current groundwater monitoring program for Dunn Field groundwater is evaluated in Section 8, and an assessment of the potential future incorporation of PDB samplers into the monitoring program is presented in Section 9. Section 10 provides a summary of the findings and decisions of the draft ROD for the MI, identifies regulatory options to be considered during the 5-year ROD review, and proposes a methodology for development of soil cleanup goals for Dunn Field. A draft CTC and STC are presented in Section 11. Section 12 summarizes recommendations for RPO opportunities and discusses their implementation. Section 14 lists the references cited in this document.

Soil gas field screening, laboratory analytical data, and SVE well construction diagrams are provided in Appendix A. Biodegradation rate calculations performed as part of the natural attenuation evaluation are presented in Appendix B. Plume-stability evaluation data, including BIOCHLOR model input and output and statistical analysis results, are contained in Appendix C. Supporting data for the Dunn Field groundwater monitoring program evaluation are provided in Appendix D, and Appendix E contains supporting data for the CTC and STC estimates and a cost estimate for a vegetable oil injection pilot test. Appendix F contains supporting data for the risk-based cleanup goal calculation described in Section 10.

SECTION 2

SITE INFORMATION

This section was prepared using all relevant information obtained prior to February 2001, and does not reflect information collected by CH2M Hill during recent site characterization events. This more-recent information, and associated revisions to conceptual site models, will be summarized in revised characterization documents prepared by CH2M Hill.

2.1 SITE HISTORY

2.1.1 Location, Operational and Regulatory History, and Background

DDMT covers 642 acres of land in the south-central section of Memphis in Shelby County, Tennessee. DDMT was established in the early 1940s. Its initial mission and function was to provide stock control, storage, and maintenance services for the Army Engineer, Chemical, and Quartermaster Corps. During World War II, the facility served as an internment center for 800 prisoners of war and performed supply missions for the Signal and Ordnance Corps. From 1963 until closure on September 30, 1997, the facility served as a major field installation for the DLA for shipping and receiving a variety of materials, including hazardous substances (pesticides, swimming pool chemicals, firearms cleaning and rust-preventative chemicals); textile products; food products; electronic equipment; construction materials; and industrial, medical, and general supplies. The Depot received, warehoused, and distributed supplies common to all US military services in the southeastern United States, Puerto Rico, and Panama. Approximately four million line items were received and shipped by the Depot annually.

DDMT is divided into two areas, the MI and Dunn Field, each with distinct infrastructure and land uses (Figure 2.1). The MI comprises 578 acres of primarily (approximately 57-percent) developed land, and includes open storage areas, warehouses, military family housing, and recreational areas. During the 1994-1995 planning stages of the Depot's Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) program, representatives of the Depot, USAESCH, United States Environmental Protection Agency (USEPA) Region 4, and the Tennessee Department of Environment and Conservation (TDEC) divided the MI into four operable units (OUs) to facilitate remediation. In 1998, after review of BRAC characterization data and consideration of the subdivision of the Depot's MI into 31 BRAC property-transfer-parcels, the MI OUs were reorganized into seven Functional Units (FUs) based on similar historical land use, potential reuse, and media of concern (Figure 2.2). The FUs at the MI are as follow:

Figure 2.1 Location of Defense Depot Memphis, Tennessee

Figure 2.2 Functional Units at the Main Installation

- FU 1- 20 Typical Warehouses;
- FU 2- Southeast Golf Course/Recreational Area;
- FU 3- Southwest Open Warehouses;
- FU 4- Northern and Central Open Areas;
- FU 5- Newer Warehouses;
- FU 6- Administrative and Residential Areas; and
- FU 7- MI Groundwater.

During the RSV, those FUs that had either completed remedial actions, had no further action planned, or had removal actions underway or planned for the near future, were eliminated from further evaluation for this RPO effort.

As shown on Figure 2.2, Dunn Field lies just north of the MI, across Dunn Avenue, and consists of approximately 64 acres of undeveloped land. About two-thirds of the area is grass covered; the remaining area is covered with crushed-rock and pavement. Dunn Field was used for bulk mineral storage and historically for waste disposal. Based on information obtained from Depot records and interviews with former Depot military personnel, disposal of hazardous and nonhazardous material as well as chemical warfare materiel (CWM) occurred at Dunn Field.

In July 1946, 29 mustard-agent-filled German bomb casings were drained, neutralized, and burned at Sites 24A and 24B, located in the southwest corner of Dunn Field. These bombs were part of a rail shipment en route from Mobile, Alabama to Pine Bluff, Arkansas. Prior to reaching Pine Bluff, three railcars were identified as containing leaking munitions and were transferred to the Memphis General Depot for proper handling. As the bombs were unloaded from the railcars, those found to be leaking were taken to a slurry pit constructed at Dunn Field for draining the mustard agent. The pit, which was reportedly 30 feet long, 7 feet wide, and 12 feet deep (CH2M Hill, 1999a), contained a chlorinated lime slurry. Reports indicate that the drained bomb casings were destroyed in a shallow trench using dynamite in the event that any of the bombs contained a burster charge.

During the early to mid-1950s, Chemical Agent Identification Sets (CAISs) were disposed of and buried at Dunn Field. The CAISs contained small glass ampules of diluted mustard agent, lewisite (a vesicant chemical agent), chloropicrin, and phosgene, which were stored in sealed cylindrical metal containers. CAIS stocks found to be leaking or broken during periodic inspection were reportedly buried at Dunn Field. The damaged CAISs may have been broken up and neutralized with chlorinated lime; however, reports indicate that on a least five or six occasions, the sets were placed into the pits intact.

The aboveground structures at Dunn Field include well heads and lift stations (associated with a groundwater extraction system that began continuous operation in

November 1998), the pistol range building, and power lines. Representatives of the Depot, USAESCH, USEPA, and TDEC designated all of Dunn Field as OU1. During the remedial investigation (RI) process, Dunn Field was divided into three geographic areas, shown on Figure 2.3, based on similar patterns of contamination, Dunn Field groundwater was designated as a separate unit. The RSV Team considered each of these four areas of concern:

- Northeast (NE) Open Area – Mowed and forested areas with limited historical waste disposal;
- Disposal Area – Location of former burial pits and trenches, including mustard-agent bomb-casing neutralization/burning areas (Sites 24A and 24B), and CWM and CAIS disposal pits (Site 1);
- Stockpile Area – Location of former aboveground storage of mineral ores and other materials; and
- Dunn Field Groundwater.

2.1.2 Previous Investigations and Remedial Actions

Table 2.1 provides a summary of investigation-related events for the MI through 1999. More recent documents have been produced that are not included in this Table. In 1981, DLA and the US Army Toxic and Hazardous Materials Agency (USATHAMA, 1982) conducted an Installation Assessment (IA) to identify historical waste disposal areas and to review waste management practices as part of the Installation Restoration Program (IRP). The IA indicated that some past waste management practices were not compatible with waste management practices in use at the time of the inquiry. The study identified areas where hazardous materials might have been used, stored, treated, or disposed of at the Depot. Based on the IA findings, USATHAMA (1982) recommended that DLA conduct a field survey.

In 1982, the US Army Environmental Hygiene Agency (USAEHA, 1982) conducted a study to characterize the geohydrologic setting and to identify and monitor sources of potential groundwater contamination. The study identified two areas of the Depot as having the potential for groundwater contamination: the MI Pentachlorophenol (PCP) Dip Vat Area and Dunn Field.

In 1985, USAEHA (1985) conducted an environmental audit of the Depot's waste management and disposal practices. The audit revealed the presence of damaged containers of acids, bases, solvents, and cleaners in the vicinity of Building 873, located in the southwestern portion of the MI. Spill areas and potentially contaminated soils also were identified adjacent to Building 873.

In 1989 and 1990, the Depot initiated an RI/FS of several known and suspected sources of contamination. The RI/FS was conducted on a site-wide basis to confirm the presence or absence of contamination, to evaluate the extent and significance of detected contamination, and to provide a scientific foundation for developing cleanup alternatives. The study was conducted in two phases, referred to as Phase I (primarily activities in

Figure 2.3 Area Designations at Dunn Field

TABLE 2.1
SUMMARY OF INVESTIGATION-RELATED EVENTS
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Year	Event/Activity	Media Investigated	Investigators
1982	Installation Assessment	General Records Search	USATHAMA
1982	Geohydrologic Study	GW, SB	AEHA
1985	Environmental Audit of Waste Management practices	SS	AEHA
1986	On-site Remedial Activities/PCP Dip Vat Investigation	SS, SB	O.H. Materials Co.
1986	Water Quality Biological Study at Fire Reservoir	SW,SD,AB	AEHA
1988	Master Plan Report	General	Harland, Bartholomew and Assoc., Inc.
1989	Storm Drainage System Maps	SW	Memphis Corps of Engineers
1990	Remedial Investigation	SS, SB, GW, SD, SW	Law Environmental, Inc.
1990	Feasibility Study	SS, SW, GW	Law Environmental, Inc.
1990	RCRA Facility Assessment	General	A.T. Kearney/EPA Region IV
1991	Pump Test	GW, SB	Engineering Science
1993	Environmental Assessment Removal Action for Groundwater	GW	Engineering Science
1993	Groundwater Monitoring	GW	ESE
1994	Focused Feasibility Study	GW	Engineering Science
1994	Lake Danielson Water Samples	SW	Pickering Environmental
1995	Work Plans and Field Sampling Plans for Main Installation and Dunn Field	GW, SW, SD, SS, SB	CH2M HILL
1995/1996	Sampling of Off-site Drainage Pathways	SW, SD	EDAW, Inc./Earth Tech
1996	Sampling of Media according to 1995 Work Plans	GW, SW, SD, SS, SB	CH2M HILL
1996	Environmental Baseline Survey	General	Woodward-Clyde, Inc.
1997	Groundwater Characterization	GW	CH2M HILL
1997	Depot Redevelopment Plan	General	Pathfinders, Inc.
1997	Sampling for Background Values in Memphis Area	SS, SB, SW, SD, GW	CH2M HILL
1998	Interim Letter Reports on 1996 sampling	SS, SB, SW, SD, GW	CH2M HILL

TABLE 2.1 (CONTINUED)
SUMMARY OF INVESTIGATION-RELATED EVENTS
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Year	Event/Activity	Media Investigated	Investigators
1998	Work Plans for Additional Sampling Prepared	SS, SB, SW, SD, GW	CH2M HILL
1998	Preliminary Risk Evaluation Performed	SS, SW	CH2M HILL
1998	BRAC Cleanup Plan Version 2 Final Report	SS, SB	Memphis Depot Caretaker
1998	Historical Aerial Photograph Analysis of Main Installation	SS	Army Topographic Engineering Center
1999	Baseline Risk Assessment for Lake Danielson and Golf Course Pond	SW, SD, AB	Radian International LLC
1999	Engineering Cost Analysis for Old Paint Shop & Maintenance Areas	SS, SB	CH2M HILL
1999	Quarterly Groundwater Monitoring Reports	GW	CH2M HILL
1999	Streamlined Risk Assessment for Parcel 3 –Technical Memorandum	SS, SB, SW, SD	CH2M HILL
1999	EE/CA Report for Parcels 28 and 35	SS	CH2M HILL
1999	No Further Action Sites Report Submitted	General	CH2M HILL
1999	Draft Final Main Installation RI Report submitted	SS, SB, SW, SD, GW	CH2M HILL
1999	Draft Feasibility Study Report Submitted	SS, GW	CH2M HILL

Notes: GW = Groundwater; N/A = Not Applicable
SW = Surface Water; AB = Aquatic Biota
SD = Sediment; PCP = Pentachlorophenol
SS = Surface Soils; BRAC = Base Realignment and Closure
SB = Subsurface Soils; RCRA = Resource Conservation and Recovery Act
EE/CA = Engineering evaluation/cost evaluation
AEHA = U.S. Army Environmental Hygiene Agency
USATHAMA = U.S. Army Toxic and Hazardous Materials Agency

Source: CH2M Hill, 2000a.

1989) and Phase II (primarily activities in 1990). The final RI report was released in April 1990 (Law Environmental, 1990a), and the final FS report was released in September 1990 (Law Environmental, 1990b). Study results indicated that the Fluvial aquifer beneath Dunn Field was contaminated, and that additional investigation was needed to fully identify contaminant source areas and to delineate the contaminant plumes.

Based on the findings of the initial RI/FS, further investigations were initiated by CH2M Hill on behalf of the US Army Corps of Engineers (USACE) to evaluate the nature and extent of contamination at the MI and Dunn Field. The results of these studies are summarized by area below.

2.1.2.1 Main Installation

Remedial Investigation and Baseline Risk Assessment. The Final RI report for the MI (CH2M Hill, 2000a) was released in January 2000. Soil analytical results for the MI indicated that soils were contaminated with lead, arsenic, polynuclear aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), tetrachlorodibenzo-p-dioxin (TCDD), and dieldrin. Analytical results for groundwater indicated the presence of volatile organic compounds (VOCs) (primarily tetrachlorethene [PCE], trichloroethene [TCE], carbon tetrachloride, and chloroform) detected in three distinct plumes in the southwestern, central, and southeastern portions of the MI. VOC concentrations were highest in the southwestern groundwater plume, where PCE was detected at a maximum concentration of 120 micrograms per liter (µg/L). Concentrations of metals in groundwater in the southwestern corner of the MI suggested that sandblasting and painting operations were sources of groundwater contamination.

The potential risks associated with industrial, recreational, and residential exposures to contaminants in soil at the MI were evaluated in the baseline risk assessment (BRA) for the MI (CH2M Hill, 2000a). Recreational land-uses were evaluated only at the southeast golf course/recreational area (FU2).

Per the BRA, it is considered unlikely that the MI will be used for residential purposes (except for the former Base housing area, which has been identified for use as transitional housing for the homeless [CH2M Hill, 2001a]) for the following reasons:

- The MI currently is zoned light-industrial;
- Depot redevelopment plans do not include future residential development except at the former Base housing area;
- The large warehouses are valuable commercial assets; and
- Industrial/commercial uses offer the potential for employment.

Therefore, residential land-uses were evaluated in the MI BRA primarily for comparison purposes only.

The BRA results showed that under current and future industrial land-use scenarios, unacceptable risks were primarily associated with future worker exposures to lead at FUs 3 and 4. However, if the MI were to be used for residential purposes in the future, primary risk drivers would include PAHs at FU 1, FU 5, and FU 6; lead and PAHs at FU 3; lead at FU 4; and arsenic and dieldrin at FU 2.

The BRA for the MI compared detected chemical concentrations in soil to soil screening levels intended to be protective of groundwater quality. No source areas of concern for chemical migration to groundwater were identified at the MI. The BRA also evaluated potential future industrial and residential exposures to contaminants in groundwater. Exposure to groundwater was evaluated for informational purposes only because there is no current or expected future use of groundwater within or surrounding the Depot. The BRA results showed elevated risks associated with industrial and residential receptor exposures to groundwater in the Fluvial aquifer underlying the Depot, primarily due to PCE and TCE. Although arsenic in groundwater also was a significant risk driver, the detected arsenic concentrations may be representative of ambient (i.e., naturally occurring) concentrations (CH2M Hill, 2000a).

Based on the BRA, preventing off-site migration of contaminated Fluvial aquifer groundwater into the Memphis Sands aquifer and the capture zone of the Allen Well Field that serves the city of Memphis (Figure 2.4), became an important factor in evaluating groundwater remedial alternatives. The Allen Well Field is located west of the Depot.

Feasibility Studies. The FS for the MI soils (FUs 1 through 6) was finalized in July 2000 (CH2M Hill, 2000e). The FS evaluated five remedial alternatives for surface soils: No Action, Limited Access, Containment, *in situ* Remediation, and Excavation and Off-Site Disposal. The FS report did not identify a preferred alternative; however, contaminated soils at several areas of the MI have been removed. As a result of the interim soil removal actions that have already taken place at the MI, FUs 1, 2, 3, and 5 have been deemed suitable for their anticipated non-residential future uses without any further action.

Soil remedial goals were developed based on industrial land-use scenarios. Per the FS, lead was the only chemical of concern (COC) identified for industrial workers. The proposed industrial cleanup goal for lead in soil (1,536 milligrams per kilogram [mg/kg]) was developed using a blood-lead uptake model for adult workers. Residential soil cleanup goals also were developed in the soils FS for informational purposes only. Per the FS (CH2M Hill, 2000e), "The Depot will not be responsible for implementing or funding remediation required to make the property available for residential use."

The FS for MI groundwater (FU 7) also was finalized in July 2000 (CH2M Hill, 2000f). As discussed further in Section 2.5, the FS evaluated five groundwater remedial alternatives for the MI: No Action; Institutional Controls/LTM; Enhanced Bioremediation; Air Sparging; and Groundwater Extraction and Discharge. The "Institutional Controls with LTM" alternative was not considered for the remediation of groundwater beneath the MI because it does not satisfy the "community acceptance" criterion under CERCLA. Instead, the option of injecting nutrients/chemicals (e.g., the REGENESIS bioremediation product HRC™ [hydrogen-releasing compound]) to

Figure 2.4 Location of Allen Well Field

promote biodegradation of CVOCs in groundwater and reduce the estimated time to achieve groundwater cleanup goals, has been selected for implementation subject to the outcome of a pilot test (CH2M Hill, 2001a). The proposed groundwater cleanup goals for PCE, TCE, and arsenic are based on current national drinking water standards (i.e., maximum contaminant levels [MCLs]).

Record of Decision. A draft ROD for the MI was released in January 2001 (CH2M Hill, 2001a), and a final ROD was signed in late February 2001. The ROD describes the selected remedies for contaminated soil and groundwater, which include the following major components:

- Excavation and off-site disposal of surface soil at FU 4 that contains lead concentrations exceeding the cleanup level of 1,536 mg/kg;
- Deed restrictions and site controls to limit residential land use;
- Enhanced bioremediation of CVOCs in the most contaminated part of the groundwater plume;
- LTM of groundwater; and
- 5-year reviews of the effectiveness of the selected alternatives.

2.1.2.2 Dunn Field

Interim Record of Decision. An interim ROD for interim remedial action for the groundwater at Dunn Field was approved on April 9, 1996. The interim ROD provided for hydraulic control of the contaminant plume in groundwater beneath Dunn Field through the installation of recovery wells along the western boundary of Dunn Field. Extracted groundwater was to be discharged to the sanitary sewer. The design for the complete 11-well groundwater recovery system was finalized in August 1997. A phased installation was planned, with the performance of the first seven wells to be used to evaluate the placement of additional wells. Installation of the first seven extraction wells and construction of the pumping system and hookups to the City of Memphis sewer system was completed in October 1998. Due to increasing concentrations of CVOCs observed in some extraction wells, four additional wells have been installed, and will be operational starting in January 2001. The interim remedial action for groundwater at Dunn Field is described in more detail in Section 2.6.

Engineering Evaluation/Cost Analysis (EE/CA). An EE/CA for the removal of chemical warfare materiel (CWM) from suspected disposal/burial pits at Dunn Field was issued in June 1999 (Parsons ES, 1999). The EE/CA was performed to assess whether CWM contamination was leaving suspected disposal/burial pits, to analyze risk management and removal alternatives, and to recommend a feasible CWM removal alternative for any contaminants present.

Laboratory analytical results for soil samples collected from the suspected burial pit areas indicated that no migration of CWM or breakdown products from the pits or trenches has occurred. A streamlined risk evaluation indicated that adverse effects to current and future human receptors resulting from exposure to site media are not expected to occur in the areas directly adjacent to the suspected CWM burial pits. However, excavation and removal of CWM was recommended because it was assumed that toxic CWM exists in the suspected burial pits.

Remedial Investigation and Baseline Risk Assessment. The Draft RI for Dunn Field (CH2M Hill, 1999a) was released in October 1999, and the Draft Final RI was submitted to the BRAC Cleanup Team (BCT) in March 2000. As described in more detail below, a final version of the RI has not been released due to unexpectedly high TCE and 1,1,2,2-tetrachloroethane (1,1,2,2-PCA) concentrations observed in offsite monitoring wells sampled in February 2000. During the RI, each of the three source areas (i.e., NE Open Area, Disposal Area, and Stockpile Area) and the groundwater beneath the entire Dunn Field area were evaluated. Soil analytical results indicated that CVOCs were the primary contaminants present in surface and subsurface soil at the NE Open Area and Disposal Area.

In December 2000, the Base Conversion Team (BCT) requested that the BRA be revised to include a re-review of the data for the eastern portion of Dunn Field for possible change in the unrestricted use status of this area.

The RI for Dunn Field identified two distinct plumes of VOC contamination in groundwater. Plume A, located in the northeastern corner of Dunn Field, primarily consists of 1,1,1-trichloroethane (1,1,1-TCA) and 1,1-dichloroethene (1,1-DCE), while Plume B, located at the western side of Dunn Field, primarily consists of PCE, TCE, 1,1,2,2-PCA, carbon tetrachloride, and various degradation products of these chemicals.

The BRA for Dunn Field (CH2M Hill, 2000b) evaluated industrial, recreational, and residential exposures to contaminants in soil at the site. Recreational land-use scenarios were evaluated only at the wooded area located in the northeastern portion of Dunn Field. As for the MI, residential land-use scenarios were evaluated for comparison purposes only. It is considered unlikely that the site will be used for residential purposes (CH2M Hill, 2000b). In December 2000, the Base Conversion Team (BCT) requested that the BRA be revised to include a re-review of the data for the eastern portion of Dunn Field for possible change in the unrestricted use status of this area.

The BRA results showed that under current and future industrial land use conditions, elevated risks primarily were associated with future worker exposures to lead at the NE Open Area firing range and to VOCs (primarily 1,1,2,2-PCA, 1,1,2-TCA, TCE, carbon tetrachloride, and vinyl chloride [VC]) released from soils into the indoor air of future buildings in the NE Open Area and the Disposal Area. However, if Dunn Field were to be used for residential purposes, then elevated risks primarily would be associated with arsenic, antimony, and PAHs in soil, and with VOCs in indoor air.

Potential future industrial and residential exposures to contaminants in groundwater also were evaluated in the BRA for Dunn Field (CH2M Hill, 2000b). Exposure to groundwater was evaluated for informational purposes only, as there is no current or expected future use of groundwater within or surrounding the Depot. The BRA results showed elevated risks associated with industrial and residential exposures to groundwater, primarily due to 1,1-DCE, 1,1,2,2-PCA, carbon tetrachloride, chloroform, PCE, and TCE.

Recent Groundwater Monitoring Results. In November 1999 and February 2000, monitoring wells were installed west of Dunn Field to provide water level data for the Fluvial aquifer and to evaluate the capture zone of the Dunn Field interim groundwater extraction system. Groundwater samples collected from these monitoring wells during the February 2000 sampling event showed TCE and 1,1,2,2-PCA to be present at concentrations one order of magnitude greater than previously detected west of (downgradient from) Dunn Field. The detection of TCE in MW70 at 11,700 µg/L (1.06% of its aqueous solubility) indicated the potential presence of dense nonaqueous-phase liquid (DNAPL), and represented a change in the contaminant delineation and the fate and transport conceptual model previously developed for the site. Analytical results for a sample collected from well MW70 in March 2000 showed TCE and 1,1,2,2-PCA at concentrations an order of magnitude lower than the February 2000 results. Based on these inconsistencies, and on the need to further investigate the extent of groundwater contamination at Dunn Field, the Draft Final RI for Dunn Field was recalled. The Final RI Field Sampling Plan Addendum II for Dunn Field (OU1) was released by CH2M Hill (2000d) in June 2000.

2.1.2.3 Natural Attenuation Evaluation

In March 2000, a natural attenuation assessment was conducted to evaluate monitored natural attenuation (MNA) as a groundwater alternative in the FSs for the MI and Dunn Field (CH2M Hill, 2000c). Groundwater samples were collected on March 21 through 24, 2000 from nine wells at the southwestern corner of the MI, and from eight wells at Dunn Field. Groundwater samples were analyzed for VOCs and a suite of natural attenuation indicator parameters.

The study results indicated limited evidence for biodegradation of CVOCs at five of the nine wells sampled at the MI and at six of the eight wells sampled at Dunn Field. The remaining wells at both sites showed inadequate evidence for biodegradation of chlorinated organics. The study concluded that some degradation of highly chlorinated solvents, primarily by the reductive dehalogenation of PCE to TCE to DCE, is occurring. The study noted that the overall dissolved oxygen (DO) measurements at both the MI and Dunn Field indicated that the Fluvial aquifer is aerobic, a condition that does not support the reductive dehalogenation process.

2.2 SITE GEOLOGY AND HYDROGEOLOGY

2.2.1 Geologic and Hydrogeologic Setting

Four geologic units underlie the Depot and influence groundwater flow and contaminant migration. These are, from shallow to deep: loess, fluvial deposits, Jackson Formation/Upper Claiborne Group, and the Memphis Sand aquifer (Figure 2.5). It should be noted that the conceptual site model (CSM) is being updated and refined as new characterization data are obtained (i.e., as a result of CH2M Hill's DNAPL investigation at Dunn Field). Therefore, the information presented in this section may be revised as new data become available.

Figure 2.5 Cross-Section Through Area of Suspected Exposures of the Confined Sand Aquifer

2.2.1.1 Loess

The unsaturated loess unit is a layer of firm silty clay or clayey silt that is approximately 6 to 40 feet thick (CH2M Hill, 2000a). Where intact and undisturbed, the loess unit tends to limit precipitation infiltration (recharge) to underlying aquifers. Sandy zones within the loess may become seasonal perched water-bearing zones that contain water for short periods of time after rainfall events. The permeability range for the soil is 4.4×10^{-4} to 1.4×10^{-3} centimeters per second (cm/sec) (1.2 to 4.0 feet per day [ft/day]) (Memphis Depot Caretaker, 1999).

2.2.1.2 Fluvial Deposits

Fluvial deposits underlie the loess. This unit is composed of three generalized members that can be identified at the site: silty clay, silty sandy clay, or clayey sand; poorly graded, fine- to medium-grained sand; and gravelly sand. The lower, saturated portion of the fluvial deposits is referred to as the Fluvial aquifer and is the uppermost unconfined aquifer beneath the Depot. Saturated thicknesses of the Fluvial aquifer are variable across the Depot and are partially controlled by the configuration of the uppermost clay in the underlying Jackson Formation/Upper Claiborne Group (Figure 2.6). Saturated thicknesses of the Fluvial aquifer at the Depot generally range between 10 and 15 feet. As discussed in more detail in Sections 2.2.2 and 2.2.3, the saturated thicknesses of the Fluvial aquifer reach a maximum range of 30 to 40 feet in the southwestern portion of Dunn Field and the northwestern portion of the MI, where the confining unit clay beneath the Fluvial aquifer may be absent or thin.

Hydraulic conductivity values from Law Environmental (1990a) and CH2M Hill (1997) were summarized in the Final RI report for the MI (CH2M Hill, 2000a). Hydraulic conductivities in the Fluvial aquifer ranged from 6.7×10^{-5} to 2.5×10^{-2} cm/sec (0.2 to 71 ft/day), with the highest values recorded near the southern and southwestern portions of the MI. The geometric mean hydraulic conductivity for wells within the MI was 2.2×10^{-3} cm/sec (6.2 ft/day). In 1992, a pump test was performed in the northwestern portion of Dunn Field (MW3) to measure hydraulic parameters needed for design of the Dunn Field groundwater extraction system. The average hydraulic conductivity value obtained via pump testing of the Fluvial aquifer was 3.4×10^{-2} cm/sec (96 ft/day), approximately one order of magnitude higher than the values obtained by slug testing (Engineering-Science, Inc., 1994.)

The magnitude of groundwater gradients in the Fluvial aquifer underlying the MI appear to be greatest in the northwestern portion of the property. A maximum groundwater velocity in this vicinity was estimated at 11.8 ft/day, based on a hydraulic gradient of 0.16 foot per foot (ft/ft), an hydraulic conductivity of 22.1 ft/day, and an effective porosity of 0.3 (CH2M Hill, 2000a). However, as discussed in Section 7.1.4, this gradient may not be representative of the Fluvial aquifer. At Dunn Field, the range for groundwater velocity was estimated at 0.1 ft/day to 1.7 ft/day, based on a hydraulic gradient ranging from 0.0017 ft/ft to 0.023 ft/ft along the western boundary of Dunn Field, an hydraulic conductivity of 22.1 ft/day, and an effective porosity of 0.3 (CH2M Hill, 1999a).

Figure 2.6 Top of the Uppermost Clay in the Jackson Formation/Upper Claiborne Group

Recharge to the Fluvial aquifer is primarily from the infiltration of rainfall (Graham and Parks, 1986). Discharge from the Fluvial aquifer is generally directed toward underlying units in hydraulic communication with the fluvial deposits, or laterally into stream channels. Within the Depot area, the Fluvial aquifer is unconfined, as the water table was consistently observed below the base of the overlying loess. The Fluvial aquifer is not used as a drinking water source.

2.2.1.3 Jackson Formation/Upper Claiborne Group

The Late Eocene Jackson Formation and upper part of the Claiborne Group lie beneath the fluvial deposits. The Upper Claiborne consists of the Jackson, Cockfield, and Cook Mountain Formations. The Jackson and Cockfield Formations consist of sand, silt, clay, and lignite beds. Because of lithologic similarities, the Jackson and the Cockfield Formations cannot reliably be subdivided in the subsurface of the Memphis area. Where present, clays within the Jackson/Cockfield Formation constitute the base of the Fluvial aquifer.

The surface topography of the uppermost clay in the Jackson Formation/Upper Claiborne Group in the area of the Depot is depicted on Figure 2.6. This unit is represented at the site by a distinctive stiff gray, low- to high-plasticity lignitic clay. Analytical results for samples collected from the top of the uppermost confining unit clay indicated that the clay has a very low permeability, with hydraulic conductivities ranging from 1.2×10^{-8} to 2.5×10^{-7} cm/sec (3.4×10^{-5} to 7.1×10^{-4} ft/day), respectively. Therefore, the uppermost clay in the Jackson Formation/Upper Claiborne Group, where present, constitutes a hydraulic barrier to downward migration of groundwater from the Fluvial aquifer to the underlying aquifers (CH2M Hill, 2000a). As discussed in more detail in Sections 2.2.2 and 2.2.3, the RI results indicate that downward leakage from the Fluvial aquifer to the underlying Memphis aquifer could potentially occur in the southwestern portion of Dunn Field and the northwestern portion of the MI (near to wells MW34 and MW38), where the confining unit clay may be absent or thin (Figures 2.5 and 2.6). Figure 2.6 suggests that the clay unit thins in this area are due to the presence of a topographic low in the clay surface, which may represent an erosion surface. In contrast, Figure 2.5 suggests that the clay may be absent in this area.

The thickness of the Jackson Formation is reported variously in literature. Kingsbury and Parks (1993) report a range of 0 to 50 feet, while Parks and Carmichael (1988) report a thickness ranging from 0 to 150 feet. Where the Jackson Formation is present, the Cockfield may range from 235 to 270 feet in thickness. In some areas, the Cockfield Formation contains sands that comprise the Cockfield aquifer. This aquifer normally is confined (see confined sand aquifer on Figure 2.5), but locally may be unconfined (Parks and Carmichael, 1988), and provides water for some public and industrial uses.

The Cook Mountain Formation is the lower confining unit to the Cockfield and generally consists of clay, silt, and sand. Kingsbury and Parks (1993) report a range of 0 to 50 feet in the Memphis area, while Parks and Carmichael (1988) report a thickness ranging from 0 to 150 feet over the west Tennessee area (CH2M Hill, 1999a). The Cook Mountain, Cockfield, and Jackson Formation sequence serves as the upper confining unit to the Memphis Sand aquifer (Figure 2.5).

2.2.1.4 Memphis Sand Aquifer

The Memphis aquifer underlies the Depot at a depth of approximately 180 feet bgs and averages 500 feet in thickness (Figure 2.5). In general, this aquifer contains groundwater under strong artesian (confined) conditions. The Memphis aquifer derives most of its recharge from areas where it crops out. The outcrop area forms a wide northeast-trending belt east of Memphis. The outcrop belt extends from east of Shelby, Fayette, and Hardeman Counties northeast across much of west Tennessee. The Memphis aquifer is the primary drinking water source for Memphis.

2.2.2 Hydrogeologic Interactions

As discussed in Section 2.2.1.3, downward leakage from the Fluvial aquifer to the underlying Memphis aquifer could potentially occur in the southwestern portion of Dunn Field and the northwestern portion of the MI (near wells MW34 and MW38), where the confining unit clay may be absent or thinning.

Section 2.7.3 of the Final RI for the MI (CH2M Hill, 2000a) provides a hydrogeologic interpretation of the interconnectiveness of the Fluvial and Memphis aquifers based on tritium data collected from Depot wells. According to the Final RI, the tritium data suggest mixing of Fluvial aquifer groundwater with the Memphis aquifer at MW36 (Figure 2.6). This interpretation is consistent with the stratigraphic interpretation of a gap in the confining unit clay observed near MW34 and STB-13, as depicted on Figure 2.5. Further investigation of the hydraulic connection between the aquifers is planned as part of ongoing investigations.

2.2.3 Groundwater Flow

The configuration of the clay within the Jackson Formation/Upper Claiborne Group strongly influences the direction of groundwater flow in the Fluvial aquifer at the Depot. Groundwater flow directions within the unconfined Fluvial aquifer, based on measurements taken in November 1998, are depicted on Figure 2.7.

In the northwestern portion Dunn Field, groundwater flows west and northwest, while in the southern portion of Dunn Field, groundwater appears to flow south and southwest toward the depression in the clay confining unit in the northwest portion of the MI (MW34 and MW38) (Figure 2.7).

In the western and southwestern portions of the MI, groundwater flow has been interpreted as northeastward, toward the depression at MW34 and MW38. An alternative interpretation of groundwater flow directions in the Fluvial aquifer at DDMT, presented in the MI ROD (CH2M Hill, 2001a), is discussed in Section 7.1.4. The revised interpretation was developed using only water levels from wells screened in the Fluvial aquifer. In the southeastern portion of the MI, groundwater flows in a northeasterly direction toward a depression in the water table near MW24 and PZ03. MW62 and MW63, both installed in November 1998, indicate a slight high in the water table elevations near the center of the MI. Because the groundwater levels in these wells are 5 to 6 feet higher than those in PZ03 and MW24, it has not been established that the groundwater from the south-central portion of the MI flows toward the water table low in

Figure 2.7 Potentiometric Surface Map of the Fluvial Aquifer

the northwestern portion of the MI. The groundwater flow direction in the Memphis aquifer is generally westward, toward the Allen Well Field (Figure 2.4), a major local pumping zone.

Area groundwater and surface water levels were compared during the RI (CH2M Hill, 2000a) to evaluate groundwater/surface water interactions at or near the Depot. Based on a generalized hydrogeologic cross-section created during the 1990 investigation by Law Environmental (1990a), groundwater elevations are lower than local stream base elevations in the vicinity of the Depot; therefore, the fluvial deposits probably do not contribute to stream flow at this location. Both Cane Creek and Nonconnah Creek are located at a higher elevation than the groundwater table and may recharge the Fluvial aquifer.

2.3 NATURE AND EXTENT OF CONTAMINATION

2.3.1 Main Installation

The nature and extent of contamination was assessed for the surface soils, subsurface soils, surface water, sediments, and groundwater across the MI. The nature and extent findings are provided in detail in the Final RI report (CH2M Hill, 2000a), and are summarized by environmental medium below.

2.3.1.1 Soils

The primary contaminants of concern in surface soil and subsurface soil identified at the MI include PAHs, PCBs, dieldrin, TCDD, arsenic and lead. The Final FS for the MI soils (CH2M Hill, 2000e) was released in July 2000. The Final FS did not identify a preferred alternative; however, contaminated soils at several areas of the MI have been or are currently being removed. As stated in the final ROD (CH2M Hill, 2001b), surface soils at Building 949 that contain lead concentrations in excess of 1,536 mg/kg will be excavated.

2.3.1.2 Surface Water

Contaminants detected in surface water at the MI included low levels of metals (arsenic and lead), pesticides (dieldrin, DDT, and DDE), and dioxins. These detections were considered likely to be attributable to suspended soil particles and were not associated with unacceptable human health risks (CH2M Hill, 2000a).

2.3.1.3 Sediment

Contaminants detected in sediment at the MI included low levels of metals (arsenic, chromium, and lead), pesticides (dieldrin, DDT, and DDE), and dioxins. These detections were not associated with unacceptable human health risks (CH2M Hill, 2000a).

Figure 2.8 Groundwater Monitoring Well Locations

2.3.1.4 Groundwater

The extent of metal, VOC, and semivolatile organic compound (SVOC) contamination in the Fluvial aquifer has been characterized across the MI (CH2M Hill, 2000a). Groundwater monitoring well locations at the MI are shown on Figure 2.8.

The primary COCs in MI groundwater are VOCs (including PCE, TCE, and 1,2-DCE) and metals. The distributions of PCE and TCE detected during the February and March 2000 sampling events are shown on Figures 2.9 and 2.10**,** respectively. The analytical results for groundwater indicated the presence of two CVOC plumes in the southwestern and southeastern portions of the MI. In addition, PCE was detected at MW47, located south of the southwest corner of the MI. The two plumes of contamination at the MI, along with temporal trends in persistent CVOC concentrations at the MI, are discussed in the following sections.

<u>Southwest CVOC Plume.</u> The persistent CVOCs detected in the southwestern plume at the MI include PCE, TCE, and 1,2-DCE. The highest VOC concentration (200 µg/L for PCE) was detected in this area. Groundwater flow is from the southwest onto the MI area, suggesting the possibility of an off-site source of PCE. As discussed in Section 32.1.6 of the Final RI (CH2M Hill, 2000a), the concentrations of PCE in groundwater samples in the southwestern portion of the MI have been increasing since the RI sampling began during the first quarter of 1996. An abandoned dry cleaning facility located southeast of PZ08 was initially believed to be a possible source of PCE contamination in this area. However, no organic compounds have been detected at PZ08, nor is the dry cleaner located hydraulically upgradient from this plume. Therefore, an unidentified off-site source of PCE to the southwest of PZ04 and MW21 continues to be a possibility. It is also possible that contamination from a source of PCE on the MI has migrated off-site along the top of the clay unit, which is observed to dip toward the south and southwest (Figure 2.6), and because of the relatively flat groundwater gradient.

Metals have been detected above background levels only in the immediate vicinity of the sandblasting and painting area in the southwestern corner of the MI. Although these exceedances are minor, they suggest a possible impact on groundwater attributable to past operations. Concentrations of these metals are at or below background levels within a short distance from the paint shop. In addition, these concentrations are decreasing with time, with the highest values observed during earlier monitoring (1996) and lowest observed during more recent monitoring events (October 1998).

<u>Southeastern VOC Plume.</u> The persistent VOCs in the southeastern portion of the MI include PCE, TCE, 1,2-DCE, carbon tetrachloride, and chloroform. Specific VOC sources correlating with the southeast plume have not been identified in the subsurface (CH2M Hill, 2000a). Groundwater flow from this area is southwest toward a northwest/southeast-trending depression in the potentiometric surface in the vicinity of PZ03 and MW24. Groundwater from this area is believed to flow to this depression, then southeastward toward the MI boundary. To date, no PCE or TCE have been detected in MW24, which is the farthest downgradient well in the south-central depression.

<u>Temporal Trends in VOC Concentrations.</u> The concentrations of persistent VOCs detected at selected MI wells were plotted over time to evaluate trends in locations with

Figure 2.9 Distribution of PCE in Groundwater at the Main Installation

Figure 2.10 Distribution of TCE in Groundwater at the Main Installation

more than one sampling event. These plots are shown on Figure 2.11. In general, groundwater levels were observed to fluctuate within a range of approximately 1 foot. Temporal trends in groundwater level fluctuations in the Fluvial aquifer beneath the MI were variable, though some patterns among wells are evident. The degree of fluctuation was typically less than 10 percent of the saturated thickness of the aquifer and did not affect the general flow directions across the MI. With the exceptions noted below, neither positive nor negative correlations of groundwater level with changes in concentration were discernable. Fluctuations in VOC concentrations were erratic, with no apparent increasing or decreasing trends.

At MW21, located in the southwest corner of the MI, groundwater samples exhibited a significant positive correlation between groundwater level and VOC concentrations. The increase in concentrations of PCE, TCE, and possibly 1,2-DCE suggests a nearby soil source is leaching an increasingly larger mass of VOCs to groundwater. However, no VOCs have been detected in soil samples collected in this area. Plume geometry during the course of the MI RI (from January 1996 through November 1998) changed only slightly because of changes in concentration.

2.3.2 Dunn Field

The nature and extent of contamination at Dunn Field were assessed for soil at the NE Open Area, the Disposal Area, and the Stockpile Area, and for groundwater beneath the entire Dunn Field area (Figure 2.3). The results of the site characterization are provided in detail in the Draft and Draft Final RI reports (CH2M Hill, 1999a and 2000b), and are summarized by environmental medium below.

2.3.2.1 NE Open Area Soils

Soil analytical results indicated high concentrations of lead in surface soil and dieldrin in subsurface soil at the pistol range at the NE Open Area. Low metal concentrations in surface soils occurred at random and isolated locations throughout the remainder of the NE Open Area. VOCs were detected in both surface soil and subsurface soil in this area, suggesting that casual surface disposal of chlorinated solvents may have occurred in the NE Open Area during the long periods of operations at Dunn Field. Pesticides, including dieldrin, DDD, and DDT, were also detected across the NE Open area, but were not associated with discrete releases from source areas within this area.

2.3.2.2 Disposal Area

Low concentrations of metals, including chromium, lead, antimony, and thallium in surface soils and subsurface soils, occurred at random and isolated locations throughout the Disposal Area. These metals do not appear to be associated with discrete releases from source areas at the Disposal Area. Pesticides were detected throughout the Disposal Area, and PAHs were detected near the railroad tracks.

A passive soil gas survey was conducted at the Disposal Area at Dunn Field in August 1998 (CH2M Hill, 1999a). Moderate to high soil gas VOC concentrations were detected in the northwestern corner of Dunn Field, indicating potential VOC contamination in the disposal areas and soils.

Figure 2.11 Temporal Trends in VOC Concentrations in Groundwater at the Main Installation

During the RI sampling effort, three CVOCs (TCE, 1,1,2,2-TCA, and VC) were detected above screening levels in surface soil, and eight CVOCs (1,1,2,2-PCA, 1,2-dichloroethane [DCA], carbon tetrachloride, chloroform, methylene chloride, PCE, TCE, and VC) were detected above screening levels in subsurface soil. The VOCs detected in subsurface soils at the Disposal Area are shown on Figure 2.12. The results of the subsurface soil sampling correlated well with the extent of the VOCs in the subsurface suggested by the passive soil gas survey. The apparent clustering of high VOC concentrations also correlates well with the historical information indicating that the disposal pits and trenches were relatively small and separate. The RI concluded that the Site 10 disposal pit (Solid Waste Burial Site) may be the largest single potential CVOC source of groundwater contamination at Dunn Field (CH2M Hill, 1999a).

2.3.2.3 Groundwater

The extent of dissolved metals, VOC, and SVOC contamination in the Fluvial aquifer has been characterized across Dunn Field (CH2M Hill, 1999a). The RI identified two distinct plumes of VOC contamination at Dunn Field: one at the northeastern corner (Plume A) and one at the western side of Dunn Field (Plume B). Metals (including aluminum, vanadium, iron, lead, beryllium, and manganese) were detected at frequencies and locations that suggest their occurrences could be related to waste-management practices at the site. In contrast to the VOC plumes that underlie Dunn Field, the locations where dissolved metals exceeded background concentrations are limited to small geographic areas, primarily in the northern and northwestern portions of Dunn Field.

Plume A. The VOCs consistently detected in groundwater beneath the northeastern corner of Dunn Field include 1,1,1-TCA, 1,1-DCE, PCE, TCE, and 1,1-DCA. The spatial distributions of 1,1,1-TCA and 1,1-DCE are shown on Figures 2.13 and 2.14, respectively. The spatial distributions of the 1,1,1-TCA and 1,1-DCE occurrences suggests a west-southwest-trending plume that originates offsite. Maximum concentrations of these two COCs were detected at well PZ02, located hydraulically upgradient from Dunn Field, indicating an offsite source. The spatial distributions of PCE, TCE, and their biodegradation daughter product 1,2-DCE also suggest that a plume may be present that trends west-southwest.

Plume B. Plume B is the larger plume at Dunn Field and also extends beyond the boundary of the installation. As discussed in the Draft Final RI report (CH2M Hill, 2000b), Plume B is likely attributable to VOCs detected in soil throughout the Disposal Area at Dunn Field. The persistent VOCs associated with Plume B on the western side of Dunn Field primarily include TCE, 1,2-DCE, PCE, carbon tetrachloride, 1,1,2,2-PCA, and 1,2-TCA. PCE was detected at its highest concentration (120 µg/L) in a groundwater sample from MW04 in Plume B.

The distributions and concentrations of TCE and 1,1,2,2-PCA are shown on Figures 2.15 and 2.16, respectively. The TCE plume encompasses the northwestern and western boundaries of Dunn Field and extends off-site to the west, northwest, and north. The 1,1,2,2-PCA plume is located along the western boundary of Dunn Field and extends off-site to the northwest. The PCE plume (not shown) is centered on the western and

Figure 2.12 VOCs in Subsurface Areas at the Disposal Area, Dunn Field

Figure 2.13 Distribution of 1,1,1-TCA in Groundwater at Dunn Field

Figure 2.14 Distribution of 1,1-DCE in Groundwater at Dunn Field

Figure 2.15 Distribution of TCE in Groundwater at Dunn Field

Figure 2.16 Distribution of 1,1,2,2-PCA in Groundwater at Dunn Field

northern boundary of Dunn Field. The carbon tetrachloride plume (also not shown) is located along the western boundary of Dunn Field and extends off-site to the west.

As discussed in Section 2.1.2.2, groundwater monitoring wells were installed west of Dunn Field in November 1999 (MW69, 70, and 71) and February 2000 (MW68) to collect water level data for the Fluvial aquifer and to evaluate the capture zone of the Dunn Field groundwater extraction system. In February 2000, groundwater samples were collected from monitoring wells MW69, 70, and 71 as part of the second year of the OM&M for the Dunn Field groundwater extraction system. As shown on Figures 2.17 and 2.18, the groundwater analytical results from MW70 indicated the presence of TCE and 1,1,2,2-PCA at concentrations of 11,700 µg/L and 4,830 µg/L, respectively. These concentrations were an order-of-magnitude greater than what has been previously detected west of Dunn Field. The detected concentrations of TCE in MW70 are 1.06 percent of the aqueous solubility, which suggests the presence of DNAPL. Previous sampling events, both on and off Dunn Field, did not indicate the presence of DNAPL in the saturated zone. The concentrations of CVOCs detected in MW70, coupled with increasing TCE and 1,1,2,2-PCA concentrations in recovery well RW-5 during the past five quarters of monitoring (TCE from 433 µg/L in February 1999 to 1,170 µg/L in February 2000, and 1,1,2,2-PCA from 11.4 µg/L in February 1999 to 3,120 µg/L in February 2000), indicate a potential mobilization of groundwater CVOCs in this area.

A groundwater sample was again collected from MW70 in March 2000 as part of the MNA sampling event conducted at the MI and Dunn Field. The detected concentrations of TCE and 1,1,2,2-PCA from the March 2000 sampling event did not corroborate the analytical results from the February 2000 sampling event, as they were an order of magnitude lower. Possible reasons for the differing concentrations of VOCs may be different purging rates and/or sample collection from different intervals within the water column. The trend of lower TCE and 1,1,2,2-PCA concentrations in samples from MW-70 has continued throughout additional sampling events conducted since March 2000.

2.4 CONCEPTUAL MODELS

2.4.1 Main Installation

The Final RI for the MI (CH2M Hill, 2000a) presents a conceptual model for potential contaminant migration pathways that considers site topography, geology, hydrology, and site-related chemicals. A conceptual model focusing on the transport of VOCs in the southwestern corner of the MI is shown on Figure 2.19. There are no known off-site sources of VOCs southwest of the MI. A potential explanation for the observed groundwater contamination at PZ-04 is that it migrated upgradient from the MI by way of discontinuous lateral transport along thin clay laminae in the unsaturated zone. Lateral migration would occur in the aqueous phase during periods of recharge within temporary perched zones above the clay laminae. Figure 2.20 expands the conceptual model into three dimensions and shows groundwater flow directions and transport pathways across the Depot. An alternative conceptual model of groundwater flow direction within the Fluvial aquifer beneath the MI, presented in the draft ROD (CH2M Hill, 2001a), is described in Section 4.

Figure 2.17 TCE Concentrations – February 2000 Quarterly Monitoring at Dunn Field

Figure 2.18 1,1,2,2-PCA Concentrations – February 2000 Quarterly Monitoring at Dunn Field

Figure 2.19 Conceptual Model: Subsurface PCE Off-Site Migration Pathways at the Main Installation

Figure 2.20 Conceptual Model: Subsurface Migration Pathways at the Main Installation

With the exception of the metals soil contamination delineated in the southwestern portion of the MI, soil borehole samples have not identified any specific contaminant sources that could be linked to the VOC contamination present in the Fluvial aquifer. The relatively low concentrations of VOCs in groundwater over most of the Depot and the absence of definitive soil sources suggest that sources are diffuse and probably are the result of past industrial activities at the Depot. Releases from these sources (on- or off-site) would have directly affected soils below and near the sources. Continuing transport processes also may result in secondary releases that could affect larger areas or additional environmental media. Transport processes that are likely to be active at the site include vertical infiltration of chemicals into the subsurface and lateral and vertical migration in groundwater (CH2M Hill, 2000a).

2.4.2 Dunn Field

The Draft Final RI for Dunn Field (CH2M Hill, 2000b) provides a conceptual site model for the Dunn Field area (Figure 2.21). The Draft Final RI indicates that the relatively high concentrations of CVOCs in groundwater associated with the western portion of Dunn Field (i.e., Plume B) can be attributed to VOCs found in soil at the Disposal Area. The groundwater VOC contamination associated with the northeastern corner of Dunn Field (i.e., Plume A) cannot be attributed to soil contamination in the northeastern portion of Dunn Field, and is most likely from an offsite source.

As discussed previously, the Draft Final RI for Dunn Field (CH2M Hill, 2000b) was withdrawn based on the results of the February 2000 groundwater monitoring event. The Final RI Field Sampling Plan Addendum II for Dunn Field (Operable Unit 1) was released by CH2M Hill (2000d) in June 2000. This workplan presents a detailed description of the additional site characterization activities that were recently performed at the west-central area of Dunn Field. The specific objectives of this additional investigation were as follow:

- Establish the nature and extent of any DNAPL and the resulting dissolved plume associated with MW70;
- Identify DNAPL sources within the soil/disposal areas on Dunn Field;
- Evaluate DNAPL and aqueous-phase transport in the vicinity of MW70; and
- Incorporate the findings from the additional field investigation into a revised Dunn Field draft final RI report.

The field activities outlined in the Final RI Field Sampling Plan Addendum II (CH2M Hill, 2000d) were performed to refine the conceptual site model.

Figure 2.21 Conceptual Model: Dunn Field

2.5 DESCRIPTION OF CURRENT REMEDIAL ACTIONS AT DUNN FIELD

2.5.1 Dunn Field Groundwater Extraction System

An interim ROD for groundwater at Dunn Field was signed in April 1996, and provided for hydraulic control of the contaminant plume in groundwater beneath Dunn Field. The groundwater interim remedial action included the following elements:

- Installing groundwater extraction wells through the groundwater plume at the western boundary of Dunn Field to control further offsite migration;

- Obtaining discharge permits for disposal of recovered groundwater into the city of Memphis municipal sewer system;

- Operating the recovery wells until contaminant concentrations are reduced to acceptable levels or until the final remedy is in place; and

- Performing chemical analysis to monitor the quality of the discharge in accordance with city discharge permit requirements.

The design for the 11-well groundwater recovery system was finalized in August 1997. A phased installation was planned, with the performance of the first seven wells used to evaluate the placement of additional wells. Installation of the first seven extraction wells and construction of the pumping system and hookups to the City of Memphis sewer system were completed in October 1998. Four additional wells have been installed, and were brought on line in January 2001. The layout of the first seven extraction wells and water conveyance piping is shown on Figure 2.22. The system became continuously operational in early November 1998.

About 42 million gallons of water were extracted from beneath Dunn Field from late October 1998 through August 1999. During that time, an estimated minimum of 12.3 pounds of VOCs were removed from the Fluvial aquifer and discharged to the sewer system for treatment.

2.5.2 Soil Excavation

In 2000, the USAESCH Ordnance and Explosive Team excavated soils at Dunn Field (CWM pits, Sites 1 and 24) in an effort to locate and remove CAISs (at Site 1) that were reportedly disposed of in the 1950s (see Section 2.1.1). The potential presence of chemical warfare substances at Dunn Field had limited the scope of previous site characterization activities, because soil borehole drilling in the former chemical warfare agent disposal areas was prohibited until these areas could be cleared. Excavated soils were visually inspected for glass ampoules and analyzed for selected constituents at a field laboratory; they were then either stockpiled for use as backfill material or segregated for offsite disposal based on the analysis results. The CWM removal effort has recently been completed.

Figure 2.22 Configuration of Dunn Field Groundwater Extraction System

SECTION 3

SOIL GAS SAMPLING RESULTS AND SVE PILOT TEST RECOMMENDATIONS

Following a review of the *Remedial Investigation Field Sampling Plan Addendum II for Dunn Field (Operable Unit 1)* (CH2M Hill, 2000d), recommendations for additional data collection related to scoping of an SVE pilot test at Dunn Field were made (Parsons ES, 2000a). The recommendations incorporated into the field investigation included collecting *in-situ* soil vapor samples using a SimulProbe™. Collection of *in-situ* soil vapor samples was recommended because it was expected that they would exhibit substantially higher (more representative) concentrations of CVOCs than conventional soil headspace measurements. The results of the Simulprobe™ sampling are presented in Section 3.1. Based on those results, additional recommendations for the SVE pilot test design were made. Those recommendations are presented in Section 3.2.

3.1 SIMULPROBE™ RESULTS

The SimulProbe™ is a modified split-spoon sampler that has the capability of collecting *in situ* soil gas samples at the same time that soil samples are being collected. The SimulProbe™ is advanced ahead of the hollow-stem augers in the same manner as a conventional split-spoon. It can be used to collect continuous soil and soil gas samples, or it can be used to sample specific intervals during advancement of a boring.

In October 2000, a total of 48 soil gas samples were collected from 7 soil boreholes for field organic vapor analyzer/flame ionization detector (OVA/FID) screening. Fourteen of the samples were submitted to Air Toxics, Ltd. in Folsom, California for quantitative analysis of VOCs using USEPA Method TO-15. Analytical results for laboratory samples collected with the SimulProbe™ (detected compounds only) are presented in Table 3.1. Complete laboratory analytical and field screening results for soil vapor sampling using the SimulProbe™ are presented in Appendix A for borings SBLCA-SB-1, SBLCA-SB-2, SBLCA-SB-3, SBLCA-SB-4, SBLCA-SB-5, SBLCA-SB-8, and SBLEE-SB-1. The approximate locations of pilot-test vapor extraction wells (VWs), vapor monitoring points (VMPs), new soil borings, and previously drilled soil borings SBLCA and SBLEE are shown on Figures 3.1 and 3.2. Figure 3.3 presents the data collected at SBLCA-SB-2, and is representative of the figures included in the Appendix A. This boring, from which the most laboratory confirmation samples were collected, appears to be located near the core of a contaminant source area, and is now the location of two SVE VWs. The SimulProbe™ data indicate that contaminant concentrations at this location increase with depth, suggesting upward diffusion of volatilized contaminants from the groundwater.

TABLE 3.1
SUMMARY OF LABORATORY ANALYTICAL RESULTS FOR SOIL GAS SAMPLES FROM DUNN FIELD
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Sampling Date	Borehole ID	Sample Depth (feet)	Units	PCE[a]	TCE[b]	cis-1,2-DCE[c]	trans1,2-DCE	Vinyl Chloride	Toluene	2-Propanol	1,1,2,2-Tetrachloroethane
10/12/00	SBLCA-SB-1	70-71.5	ppbv[d]	5,200	600,000	11,000	<8,400[e]	<2,100	9,300	46,000	<2,100
10/16/00	SBLCA-SB-2	29-30.5	ppbv	16,000	100,000	1,400	<2,100	1,300	<530	<2,100	8,400
10/16/00	SBLCA-SB-2	39-40.5	ppbv	31,000	640,000	14,000	14,000	4,100	<2,700	<11,000	65,000
10/16/00	SBLCA-SB-2	49-50.5	ppbv	27,000	2,000,000	64,000	<42,000	<10,000	<10,000	<42,000	120,000
10/16/00	SBLCA-SB-2	59-60.5	ppbv	20,000	1,600,000	49,000	<40,000	<10,000	11,000	<40,000	120,000
10/16/00	SBLCA-SB-2	69-70.5	ppbv	20,000	1,800,000	50,000	<35,000	<8,800	9,800	<35,000	240,000
10/17/00	SBLCA-SB-3	69-70.5	ppbv	5,600	640,000	24,000	<12,000	<2,900	4,600	19,000	32,000
10/18/00	SBLCA-SB-4	59-60.5	ppbv	12,000	1,000,000	35,000	18,000	<4,500	4,500	<18,000	<4,500
10/19/00	SBLCA-SB-4	69-70.5	ppbv	6,800	610,000	22,000	<12,000	<3,000	4,400	<12,000	11,000
10/24/00	SBLCA-SB-5	69-70.5	ppbv	11,000	850,000	22,000	<20,000	<4,900	10,000	<20,000	29,000
10/25/00	SBLCA-SB-8	59-60.5	ppbv	16,000	1,800,000	53,000	<33,000	<8,200	<8,200	<33,000	<8,200
10/25/00	SBLCA-SB-8	69-70.5	ppbv	29,000	2,300,000	52,000	<39,000	<9,700	12,000	58,000	20,000
10/26/00	SBLEE-SB-1	19-20.5	ppbv	<16,000	640,000	950,000	460,000	3,900,000	<16,000	<65,000	<16,000
10/26/00	SBLEE-SB-1	19-20.5DUP[f]	ppbv	<3,400	600,000	880,000	420,000	3,500,000 E[g]	5,800	<14,000	5,300

[a] PCE = Tetrachloroethene.
[b] TCE = Trichloroethene.
[c] DCE = Dichloroethene.
[d] ppbv = parts per billion, volume per volume.
[e] "<" = Analyte was not detected above the indicated reporting limit.
[f] DUP = Duplicate of preceding sample.
[g] E = Exceeds instrument calibration range.

Figure 3.1 Locations of Soil Borings and SVE Pilot Test Wells at Dunn Field

Figure 3.2 SVE Vent Wells, Vapor Monitoring Points, and Soil Borings Near SBLCA

Figure 3.3 Total Volatile Organic Compound Concentrations at SBLCA-SB-2

As indicated by the soil vapor concentration-versus-depth graphs in Appendix A, there was an overall trend of increasing contaminant concentrations with depth in each of the borings drilled near SBLCA. However, some deviations from this trend occurred at borings SBLCA-SB-3 and SBLCA-SB-4. The data for boring SBLCA-SB-1, located approximately 70 feet southeast of SBLCA, suggest that the elevated soil vapor concentrations detected below a depth of 60 feet bgs may be due primarily to upward migration of volatilized contaminants from the groundwater, encountered at approximately 75 feet bgs.

Data for boring SBLEE-SB-1, drilled approximately 600 feet north of SBLCA (Appendix A), indicate that vadose zone contamination at this location resides primarily in the loess overlying the fluvial sands. As shown in Table 3.1, the soil vapor sample collected from a depth of 19 to 20.5 feet bgs from this boring contained a relatively elevated concentration of VC, suggesting that substantial reductive dehalogenation of more highly chlorinated solvents (e.g., TCE and DCE) has occurred.

The loess soil has a relatively low permeability and consists of silty clay with some sand and gravel. At most locations, the loess is cohesive and somewhat plastic. This soil type is often considered to be a poor candidate for SVE without some form of soil fracturing to improve the secondary porosity and open channels for air-flow. SVE pilot testing is required to evaluate the effectiveness of SVE in the loess. The sand that extends from the base of the loess to the water table is fine- to coarse-grained, with some gravel and occasional small cobbles. Layers of gravel and small pebbles are present at some locations. The grain size and composition of the sand appears to vary in layers throughout the vadose zone, with some relatively abrupt changes in grain size and sorting occurring between layers. The sandy soil at this site is very well suited for SVE.

3.2 SVE PILOT TEST RECOMMENDATIONS

This section presents the recommended approach and procedures for SVE pilot testing at Dunn Field at DDMT. The purpose of a pilot test would be to determine the feasibility and economics of operating a full-scale SVE system to remediate VOCs in the vadose zone near RI soil boring SBLCA at Dunn Field. Upon completion of a pilot test, the information gained could be used to develop the most effective long-term strategy for contaminant mass removal in the vadose zone at Dunn Field.

3.2.1 Objectives

The proposed objectives of an SVE pilot test at Dunn Field are to:

1. Evaluate the effectiveness of SVE by testing the existing VWs installed near the “hot-spot” identified at soil boring SBLCA-SB-2 (Figure 3.2). The main component of the effectiveness evaluation should be an examination of the contaminant mass recovery potential of the technology. In addition, the potential for preventing the downward migration of contamination through the unsaturated zone and mitigating the continuing impact to groundwater should be examined.

2. Determine design parameters (e.g., flow-rate, vacuum requirements, off-gas treatment requirements, and radius of influence) to aid in development of a conceptual full-scale SVE system design for the vadose zone. Development of a conceptual full-scale design could facilitate an evaluation of how efficient/cost-effective SVE would be at removing contaminant mass as compared to alternative technologies.

3.2.2 Overall Approach

As shown on Figure 3.2, two nested VWs were installed in boring SBLCA-SB-2, located approximately 12 feet north of the "hot spot" previously identified at soil boring SBLCA during the RI (CH2M Hill, 2000b). The shallow well, VW-1, is screened from 9 to 24 feet bgs, entirely within the loess formation. The deep well, VW-2, is screened from 32 to 72 feet bgs, entirely within the fluvial sand formation. The construction diagrams for these VWs are included in Appendix A. These two VWs were constructed to be the test wells during a pilot test, and also could be used as part of a full-scale SVE system. The depth to groundwater at the VWs at the time of installation was approximately 76 feet bgs.

Four multi-depth VMPs were installed in borings SLBCA-SB-1, SLBCA-SB-3, SLBCA-SB-5 and SLBCA-SB-8; these monitoring points are referred to as MP-1, MP-2, MP-3 and MP-4. Each VMP consists of four screened intervals within the same borehole separated by bentonite seals. The screen and sand pack intervals for the VMPs are presented in Table 3.2. The construction diagrams for these VMPs are included in Appendix A.

SVE pilot tests should be performed separately for VW-1 and VW-2; but the two tests can be conducted during the same mobilization. During each pilot test, subsurface pressures and soil gas parameters should be monitored at the four screened intervals of each of the VMPs and at a background location (see Section 3.2.3). Monitoring the pilot test blower system, subsurface pressure distribution, and soil gas parameters will aid in determining SVE system design parameters for the loess and the fluvial sand, including:

- Permeability,
- Radius of influence,

TABLE 3.2
VAPOR MONITORING POINT CONSTRUCTION DETAILS
REMEDIAL PROCESS OPTIMIZATION
MEMPHIS DEPOT, TENNESSEE

VMP Interval	MP-1 (SLBCA-SB-1)		MP-2 (SLBCA-SB-3)		MP-3 (SLBCA-SB-5)		MP-4 (SLBCA-SB-8)	
	Screen	Sand Pack	Screen	Sand Pack	Screen	Sand Pack	Screen	Sand Pack
MP-1-A	21.5-22	20-23	21.5-22	20-23	10.5-11	9.5-12	18.5-19	17.5-20
MP-1-B	34.5-35	33-37	34.5-35	33-37	26.5-27	25.5-28	47.5-48	46.5-49
MP-1-C	57.5-58	56-60	57.5-58	56-60	48.5-49	47.5-50	57.5-58	56.5-59
MP-1-D	70-70.5	68.5-70.5	70-70.5	68.5-70.5	67-67.5	66-68.5	69.5-70	68.5-71

Note: All depths are in feet below ground surface.

- Flow rates,
- Blower size and electrical requirements,
- Mass removal rates, and
- Extracted vapor treatment requirements.

These parameters will aid in determining the effectiveness of installing a full-scale SVE system and in the design of such a system.

3.2.3 Additional Vapor Monitoring Points

Installation of up to three additional VMPs is recommended prior to pilot testing. One of the additional points should be installed as a background monitoring point outside the expected radius of influence of VW-2 (at least 200 feet from the extraction well) to measure barometric-pressure effects during the pilot test. However, a groundwater monitoring well screened above the water table could be temporarily modified to supply these data. MW13 is approximately 200 feet from VW-1 and VW-2 and is screened from 66 to 81 feet bgs. This leaves 10 feet of screen above the water table (76 feet bgs) encountered while advancing the soil borings, sufficient for obtaining background barometric pressures. This well or another well could be temporarily modified with a flexible coupler, a reducing plug, and a valve to collect the background readings during the pilot test and eliminate the need for installing one new VMP.

The other two additional VMPs would be used to monitor vacuum response in and below the loess during the VW-1 pilot test. Because the VMP nearest to the VWs is located at a distance of about 40 feet, and because the radius of influence of VW-1 may be less than 40 feet, two additional VMPs should be installed closer to the VWs to properly monitor the site prior to performing the pilot tests. The VMPs should be located between VW-1 and MP-3 (Figure 3.2), and should be spaced approximately 10 and 20 feet from VW-1. The new VMP closest to VW-1 should be constructed with screens centered at depths of approximately 10 and 20 feet bgs. The second VMP should have screens centered at depths of approximately 10, 20, 50, and 70 feet bgs. The construction of the new VMPs should be similar to that of the existing VMPs (Appendix A).

3.2.4 Vacuum Blower, Piping, and Instrumentation

The SVE system recommended for use during the pilot test is diagrammed on Figure 3.4. The system consists of a vacuum blower, a knockout pot, an air filter, flow control and ambient air bleed valves, flow and temperature monitoring ports, and air sampling ports for air samples and pressure/vacuum measurements. To create sufficient vacuum in the soils, the blower should have the capability to extract approximately 200 standard cubic feet per minute (scfm) at a vacuum of approximately 100 to 200 inches of water.

Because it is important to keep moisture and particulates out of the blower, a knockout pot (to remove condensate) and an air filter (to remove particulates) should be placed in-line between the VWs and the blower. The knockout pot should have a drain and a level

Figure 3.4 Recommended Pilot Test Vapor Extraction System

sight gauge. If possible, the header pipe should be sloped back to the vent well to drain condensate back into the well.

The pipe and fittings connecting the VWs to the knockout pot can consist of 2-inch-diameter, Schedule 40 polyvinyl chloride (PVC). Flow-monitoring devices, valves, sampling ports (for sampling and/or pressure/vacuum measurements), and a vacuum-relief valve should be placed as illustrated on Figure 3.4. In addition, vapor temperatures should be monitored at on in-line temperature gauge.

3.2.5 Treatment of Extracted Vapor

Before completing the design of the pilot test system, the need to treat the extracted vapors prior to discharge to the atmosphere should be evaluated. Per Section 1200-3-9.04(4) Subpart(d) Part 24 of the TDEC Air Regulations, treatment is required if atmospheric loading exceeds 1 pound of VOCs per hour. This corresponds to a discharge concentration of 330 parts per million by volume (ppmv) total VOCs (assuming a flow rate of 200 scfm at standard temperature and pressure). Actual emissions concentrations could exceed this level based on the SimulProbe™ results for SBLCA-SB-2. The Memphis Public Health Department should be contacted for a permit or waiver (NPL exclusion) prior to commencing the pilot test. Atmospheric loading rates can be reduced by using 55-gallon vapor phase granular activated carbon (GAC) canisters to treat the discharge. Health-and-safety monitoring during the pilot test will ensure that worker health is protected.

3.2.6 Pilot Test Operations and Monitoring

3.2.6.1 Testing Operations

The intent of the pilot test would be to measure VOC concentrations in vapors from the two VWs while monitoring the subsurface pressure distribution and soil gas chemistry changes at the VMPs. This would be accomplished by monitoring vacuum, vapor flow, and soil-gas chemistry (oxygen [O_2], carbon dioxide [CO_2], total VOCs, and speciated VOCs) in soil gas extracted from the VWs. Monitoring of treated air emissions should be used to evaluate the VOC removal efficiency of the GAC units, and would ensure compliance with air emission standards during the test.

Prior to conducting the pilot test, static pressures and soil gas concentrations of O_2, CO_2, and total VOCs should be measured with direct-reading field instruments at all VW and VMP screened intervals. In addition, soil gas samples for laboratory VOC analysis should be collected from each VW. These data would represent baseline conditions for the test.

Two pilot tests should be conducted. An initial system check to ensure proper operation of mechanical equipment and vacuum and temperature gauges, to measure the initial vacuum responses, and to check airflow rates should be performed. After system adjustments, the pilot test at VW-1 should be performed to measure responses at the VMPs for the extraction vacuum and flow. The test should continue until steady-state vacuum response conditions are reached at the VMPs and approximately 5 pore-volumes of soil gas have been removed from the soils in the pilot test area (estimated to be approximately 2 days, Table 3.3). Removing multiple pore-volumes of soil gas will give an indication of realistic VOC mass removal rates. Once steady-state conditions have been reached and five pore-volumes have been extracted, the test should be continued for another 3 days at varying flow rates (e.g., 25, 50, and 100 scfm). This will help determine how flow rate/applied vacuum affects the VW-1 radius of influence. With this information, the number of extraction wells can be optimized.

The test on VW-1 should be performed first because this well is screened in the less-permeable loess formation. The radius of influence in the loess is expected to be substantially smaller than that of VW-2 in the underlying fluvial sands, and this test should have a minimal effect on VOC vapors in the sand formation. Because the radius of influence is expected to be relatively small in the loess formation, installation of two additional VMPs between VW-1 and MP-3 is recommended, as described in Section 3.2.3.

After completing the pilot test on VW-1, the pressure and O_2/CO_2/VOC concentrations in the loess should be allowed to equilibrate. After a minimum of 12 hours following testing at VW-1, another round of soil-gas sampling should be performed for the previously described parameters.

TABLE 3.3
PILOT TEST PURGE TIME ESTIMATES
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

	VW-1	VW-2
Top of Contaminated Layer (ft bgs)	8	28
Bottom of Contaminated Layer (ft bgs)	28	78
Estimated Radius of Influence (ft)	40	80
Estimated Air-Filled Void Fraction[a/]	0.35	0.35
Flow Rate (scfm)	75	200
Thickness of Treatment Zone (ft)	20	50
Volume of Soil to be Treated(ft^3)	100,531	1,005,310
Volume of Air-Filled Void (ft^3)	35,186	351,858
Five Times Air-Filled Void (ft^3)	175,929	1,759,292
Extraction Time (days)	1.6	6.1

[a/] Air-filled void fraction does not include pore space filled with normal soil moisture.

The pilot test at VW-2 should be performed to measure responses at the VMPs for the extraction vacuum and flow in the same manner as described for the pilot test on VW-1. The VW-2 test should continue until steady-state vacuum response conditions are reached at the VMPs, and approximately 5 pore-volumes of soil gas have been removed from the soils in the pilot test area (estimated to be approximately 6 days, Table 3.3). Once steady-state conditions have been reached and five pore-volumes have been extracted, the test should be continued for another 3 days at varying flow rates (e.g., 50, 100, and 200 scfm). This will help determine how flow rate/applied vacuum affects the radius of influence. With this information, the number of extraction wells can be balanced with equipment costs to minimize capital expenditure. The expected radius of influence for VW-2 is greater than that of VW-1 due to the higher permeability of the fluvial sands and the capping effect of the overlying loess. The estimated time duration for both tests combined (VW-1 and VW-2) is between 10 and 15 days.

3.2.6.2 System Monitoring

Soil vapor quality should be monitored throughout the tests in accordance with the following schedule:

- Prior to testing, baseline samples should be collected from the two VWs using SUMMA™ canisters for analysis of VOCs (USEPA Method TO-15). Total VOCs also should be measured in the field using an OVA.

- Prior to testing, all VMPs and a background location (e.g., MW13) should be sampled for static pressures, O_2, CO_2, and total VOCs using field instruments.

- During the pilot test, samples for total VOCs should be collected in Tedlar® bags and analyzed with an OVA hourly for the first 4 hours from both VWs and the GAC canister exhaust (if vapor treatment is performed). Sampling frequencies should decrease to once every 2 hours for the next 4 hours, then once every 4 to 12 hours, for the remainder of the test (e.g., every 4 hours during the day and every 12 hours at night). VOC samples should be collected for laboratory analysis at each extraction well during extraction operation at 4 hours, 12 hours, 36 hours, and 72 hours into each test, and at the end of each test using SUMMA™ canisters. These samples should be analyzed at a fixed-based laboratory for speciated VOCs using Method TO-15.

- During the pilot test, all VMPs and a background location should be sampled for pressures, O_2, CO_2, and total VOCs hourly for the first 4 hours, then at the frequency discussed above for the extraction wells for the remainder of the test. All sampling and analyses should be performed with field instruments.

- The volume of any condensate collecting in the knockout pot should be measured daily, and the knockout pot should be drained.

- In addition to the primary vapor samples specified above, two duplicate vapor samples, one ambient air blank, and one equipment blank should be collected in SUMMA™ canisters and analyzed for speciated VOCs using USEPA Method TO-15.

SECTION 4

DECISION TREE DEVELOPMENT

Decision trees are useful tools for guiding remediation projects that may take a number of years to complete. The decision-tree process allows for modifications to be made during remedial operations that may enhance the effectiveness of the systems or reduce overall costs by using feedback loops and lessons learned during various stages of the project. Decision trees were developed for: 1) groundwater monitoring program optimization; 2) groundwater extraction system shutdown; and 3) SVE system shutdown. The objective of the decision trees is to aid in future decision making, while moving toward the ultimate goal of site cleanup.

4.1 GROUNDWATER MONITORING

The development of an effective groundwater monitoring program involves locating monitoring points within a network and developing a site-specific strategy for groundwater sampling and analysis to maximize the amount of relevant information that can be obtained while minimizing costs. Groundwater monitoring programs generally have at least one of the following objectives:

- To characterize the nature and extent of contamination so that the risk to potential receptors can be assessed and appropriate remedial measures can be developed, and/or

- To monitor the performance of a remedial action in meeting remedial goals and mitigating risk to potential receptors.

The effectiveness of a monitoring program in achieving the objectives is generally evaluated qualitatively using professional judgment. In addition, statistical techniques are often used to perform temporal and spatial analyses to assist with the evaluation. Statistical methods are invaluable tools in that they provide an objective view of the data, whereas a qualitative evaluation alone is more subjective.

A decision process for developing and evaluating monitoring programs is presented in this subsection and is illustrated on Figure 4.1. The first step of the decision process is to conduct a review of site information and perform temporal and spatial analyses using qualitative and/or statistical techniques. This information is then used to assess whether or not an existing well, or a new well being considered for installation, should be included in a monitoring network. Also, this information is used to select appropriate sampling frequencies. The following subsections describe what is involved in the review of site information (Section 4.1.1), a temporal and spatial analysis (Section 4.1.2),

Figure 4.1 Decision Tree for Evaluating a Groundwater Monitoring Program

developing or evaluating a monitoring network (Section 4.1.3), and evaluating sampling frequency (Section 4.1.4).

4.1.1 Review of Site Information

Generally, the data needs for site characterization efforts differ from the data needs for evaluating the performance of remedial actions. During site characterization, when very little is known about the site, a relatively large amount of data is collected to identify the source(s) and types of groundwater contamination, the horizontal and vertical extent of the contaminant plume(s), and the potential for the plume(s) to expand and migrate over time. Once characterization is complete, additional data may need to be collected to support the development of remedial alternatives. During the period of remedial operations, which often extends over a number of years, it is important to periodically reassess the monitoring network to be sure that redundant monitoring is not occurring as a result of sampling at wells installed solely for site characterization or during the initial phases of remedial operations. Therefore, an important first step of a monitoring program evaluation is to define the overall monitoring program objectives.

Once the monitoring program objectives have been established, it is important to review relevant site information such as hydrostratigraphy, groundwater flow direction and rate, groundwater geochemistry, plume boundaries, well-completion details, etc. (see Figure 4.1) to gain an understanding of the groundwater system that is being monitored and factors that are influencing it. Monitoring objectives for each well or group of wells in the network should be established so that the importance of each well can be evaluated with respect to its monitoring purpose. Some examples of monitoring objectives for wells (or groups of wells) are as follow:

- To establish upgradient water quality;
- To define the vertical or horizontal extent of a plume and whether or not the plume is stable, expanding, or receding;
- To evaluate the performance of a remedial system;
- To detect potential bypass of contaminants through or around a remedial system; and
- To monitor a point of compliance (POC) or potential receptor exposure point.

It is useful to conduct temporal and spatial analyses along with the site information review to gain a better understanding of temporal trends, plume dynamics, and the spatial importance of monitoring wells. Methods for conducting temporal and spatial analyses are described in Section 4.1.2.

4.1.2 Temporal and Spatial Analyses

A temporal analysis is the review of chemical concentrations measured at the same point in an aquifer at different times, whereas a spatial analysis is the review of chemical concentrations measured at different points in the aquifer (laterally and vertically) at the same time. Temporal and spatial data can be examined either visually (qualitatively) or statistically. Although a visual (e.g., graphical) approach can be a very useful and quick method of reviewing data, it is sometimes appropriate and helpful to use a statistical approach for a more objective assessment.

4.1.2.1 Temporal Analysis

A visual temporal analysis involves a review of chemical data presented in the form of tables or graphs, collected from various sampling events over time, and qualitatively assessing whether or not a trend exists in the data. The importance of a trend or lack of trend depends on the monitoring objective and the location of the well. For example an increasing trend in concentrations at the toe of a plume may be an indication of plume expansion, and thus would be considered important information. On the other hand, an increasing trend inside of a remedial capture zone may be caused by shifting of “hot spots” resulting from the modification of groundwater flow paths caused by an extraction well. An increasing trend in this situation may be considered less important.

A number of statistical methods have been developed for evaluating temporal trends in chemical concentrations. The results of a statistical analysis will establish whether or not a temporal trend (increasing or decreasing) exists in a data set for a particular well at a specified confidence limit. The Mann-Kendall test (Gibbons, 1994) is a common method for analyzing trends in groundwater data. This test was used to evaluate monitoring data at the MI, as discussed in Section 7.2.

4.1.2.2 Spatial Analysis

A visual spatial analysis simply involves a review of the lateral and vertical distribution of monitoring points relative to a contaminant plume and remedial systems using maps and cross-sections, then using professional judgment to determine if there is redundancy in monitoring points or if data gaps exist. Statistical techniques can also be applied for a more objective assessment of potential redundancy and data gaps. An example of a statistical approach that can be used to select optimal locations of monitoring points in a network is described below.

Geostatistics, or the theory of regionalized variables (Clark, 1987; Rock 1988; American Society of Civil Engineers [ASCE], 1990a and 1990b), is based on the premise that the values of a variable (e.g., chemical concentrations) measured at two locations that are spatially "close together" will be more similar than values of that variable measured at two locations that are "far apart". If known sample values are used, the value of the variable (e.g., chemical concentrations) at any point within the sampled region can be estimated using a process known as "kriging" (Clark, 1987; ASCE, 1990a and 1990b). An additional advantage of kriging as an estimation technique is that the standard deviations (“errors”) associated with the values estimated at each point in the spatial domain also are calculated during the kriging process.

Areas containing estimated concentration values with elevated standard deviations represent locations where additional information could be collected to reduce uncertainties regarding the magnitude and extent of contaminants in the subsurface. This observation implies that the monitoring program could be optimized by using available information to identify those areas having the greatest associated uncertainty. Conversely, sampling points can be successively eliminated from simulations, and the standard deviations examined, to evaluate if significant loss of information (represented by increases in standard deviations) occurs as the number of sampling points is reduced. Iterative application of geostatistical estimating techniques, using tentatively identified sampling locations, can then be used to generate a sampling program that would provide an acceptable level of uncertainty regarding chemical distribution across the area to be monitored, with the minimum possible number of samples collected.

Due to the objectivity of this statistical approach, it is important to incorporate knowledge of the site hydrogeologic and contaminant conditions into the interpretation of the results. For example, the kriging approach cannot take into consideration the presence of preferential contaminant flow paths, such as paleochannels and fractures, or flow barriers, such as impermeable valley walls. Also, the vertical migration of contaminants cannot be easily represented using kriging techniques. Thus, it is important to view the statistical spatial evaluation as a tool for assisting in evaluating the monitoring program.

4.1.3 Evaluating the Monitoring Network

The importance of each well in the monitoring network is evaluated based on professional judgment considering a number of factors, including monitoring objectives, site-specific conditions, and the results of temporal-trend, and spatial analyses. Examples of reasons to include or exclude a well in a monitoring network are listed on Figure 4.1. These examples are further discussed below.

Site characterization efforts often require the installation and sampling of a number of wells to define the nature and extent of contamination. This information generally is used to assess potential risks associated with groundwater contamination and to develop a remedial solution. During these efforts, wells need to be strategically placed to define the lateral and vertical boundaries of the plume and to identify preferential pathways of contaminant migration.

By the time remedial actions are implemented at a site, the boundaries of the plume generally have been delineated. If this is the case, the number of wells used to define the extent of the plume can be reduced to a minimum number that would provide adequate information to assess plume evolution over time. Temporal analyses could be used to identify wells with increasing trends that may be indicating that the plume is expanding at that location. The upgradient and lateral boundaries of the plume are less likely to change over time compared to the downgradient boundary (assuming sufficient hydraulic gradient in a dominant flow direction), and hence would require fewer monitoring points. A spatial analysis can be used to assist in selecting wells that would be most strategic in monitoring plume boundaries.

To assess whether a plume is migrating, remaining stable, or receding in size, wells located at the toe of the plume and farther downgradient are monitored. If there is a high level of confidence that a remedial system is containing a plume, such as an extraction or injection system that effectively reverses the hydraulic gradient in the area of the plume, then fewer monitoring points may be required downgradient to monitor potential bypass of contaminants through or around the remedial system. Temporal analyses can be helpful in supporting conclusions that a plume is stable or receding in size due to a remedial system or natural attenuation. If there is sufficient evidence that a plume is stable or receding, then fewer monitoring points would be needed downgradient from the plume.

To evaluate the performance of a remedial action (e.g., groundwater extraction, *in situ* bioremediation, natural attenuation), wells are monitored at various locations within the treatment zone to assess whether the plume concentrations are decreasing with time. Wells located at the plume boundaries also can be used to demonstrate if plume recession is occurring due to remedial activities. Wells located in the remedial zone with the highest concentrations would be more useful for monitoring the progress of remediation (i.e., mass removal) over time than would wells with lower concentrations. A temporal analysis would be helpful in identifying wells within the plume that historically have contained the highest contaminant concentrations.

Monitoring water quality at a POC or a potential receptor exposure point is mainly for confirmation that contamination has not reached that point. The number of monitoring points needed for this confirmation should be small if the extent of contamination and groundwater flow paths are well documented, and an effective remedial strategy is in place.

To evaluate the importance of a well for establishing background water quality conditions upgradient of a particular contaminant source or plume, it is useful to look at temporal trends. For example if concentrations of a particular COC in samples from a background well have been below the laboratory detection limit for a number of years, it may be reasonable to conclude that the ambient or upgradient groundwater is uncontaminated and it would be appropriate to exclude the well from the monitoring program. On the other hand, if variable low levels of contamination have been detected in an upgradient well, it may be useful to include the well in the monitoring program to document the presence of background or upgradient contamination.

A well might be excluded from a monitoring network if it is located too far from the plume to provide useful monitoring data, it is often dry and does not consistently yield samples, or if it is providing information redundant to that provided by neighboring wells based on a spatial analysis. It also may be appropriate to exclude wells from monitoring if COCs in samples from the well have consistently been below laboratory detection limits or cleanup goals, and are expected to remain so in the future, or if the well is located outside of a well-established capture zone where water quality is not expected to be impacted by future plume migration. It may be appropriate to sample these types of wells less frequently in lieu of excluding them from the monitoring network. See Section 4.1.4 for additional discussion on evaluating sampling frequency.

The decision to permanently abandon an existing well that has been excluded from the monitoring program should be made on a site-specific basis. Even though a well may not be part of a current monitoring program, it may provide useful future information for preparing the site for closure after remediation objectives have been achieved. Therefore, it is recommended that existing wells that have been excluded from the monitoring program be left intact unless (1) they are damaged, (2) they need to be removed for construction purposes, (3) they do not yield representative water quality data, or (4) there is a high level of confidence that they will not be needed in the future.

4.1.4 Evaluating Sampling Frequency

The selection of sampling frequency should be part of the decision process described above for electing to include or exclude wells from a monitoring network. Figure 4.1 lists examples of general criteria to consider for selecting sampling frequency. Because the selection of an actual sampling frequency (e.g., quarterly, annually) is based on many site-specific factors, the criteria are listed with respect to relative sampling frequencies (i.e., more frequent versus less frequent). The various criteria in Figure 4.1 are discussed below.

In general, more frequent sampling is appropriate in aquifers with higher groundwater velocities (e.g., clean sands and gravels, highly-fractured rock, karst, high-porosity sandstones) than aquifers with low groundwater velocities (e.g., silts, clays, low-porosity consolidated rock). A dissolved COC could conceivably travel 1 to 10 feet per day in a typical clean fluvial deposit, thus relatively frequent sampling may be required to detect plume migration. A plume may travel only 10 feet per year in a silty, clayey deposit and would require relatively infrequent monitoring.

If a change in concentration at a well would not significantly alter the current course of action at a site, then a relatively low sampling frequency should be considered for that well. For example, changes in concentrations in wells located inside an extraction well capture zone likely will not provide a reason to modify operations for many years, thus a relatively low sampling frequency may be appropriate for at least some of the wells inside the capture zone. On the other hand, if contaminant concentrations increase at a well located outside of the capture zone, the system may need to be modified to include capture of contaminants at that location by increasing the extraction rate or adding another extraction well. Thus, more frequent sampling may be appropriate for this type of well.

If the purpose of a well is to monitor a potential release from a source area or the performance of a remedial system, then wells closer to the source or remedial system should be monitored more frequently than wells located farther downgradient. This is because a change in concentration due to a source release or due to remediation would likely be observed first in the wells closer to the source/remedial system. Changes at these wells may trigger more frequent sampling in the downgradient wells, where the change would be expected to occur at a later time.

If concentrations are expected to be relatively stable in a particular well over time, then a relatively low sampling frequency may be appropriate for that well. Some examples of wells in this category include: (1) upgradient wells that monitor background water quality, (2) wells located outside of a well-established capture zone where there is a high level of confidence that the plume is contained, and (3) wells located downgradient from a plume where it has been demonstrated that the plume is stable or receding due to natural attenuation.

4.2 GROUNDWATER EXTRACTION SYSTEM EVALUATION

Some of the key decisions that are made during the operation of a groundwater extraction system are (1) whether or not the system is effective and efficient in meeting remedial goals, (2) whether the effectiveness can be enhanced or the cost reduced by modifying the system or implementing an alternative remedy, (3) determining when remedial goals have been met or if they can they be met, and (4) determining when the system (or a portion of the system) can be shut down. A decision tree was developed to aid in making these decisions while maintaining progress toward meeting the remedial objectives for the Memphis Depot (Figure 4.2).

Generally, the goal of a groundwater extraction system is to remove enough contaminant mass from the groundwater that cleanup levels in the aquifer can be maintained after the system is shut down. For this to be achieved, a significant portion of the source has to be removed. In many cases, the source(s) cannot be adequately identified and removed or remediated, and hence cleanup goals for the aquifer cannot be met in a reasonable time frame. For example, if NAPLs are present and serving as a source of dissolved contamination, it may not be feasible to locate and remove a significant amount (>95 percent) of NAPL to provide for effective, permanent reduction in contaminant levels using groundwater extraction. In these cases, the primary objective of the groundwater extraction system would be to contain the source (i.e., limit NAPL migration), control plume migration, and reduce risks to potential downgradient receptors. Therefore, an important first step in the decision-making process is to characterize and remediate (to the extent feasible) the contaminant source and define the objective of the groundwater extraction system.

The primary goal of the current groundwater extraction system is to provide a hydraulic barrier to prevent contaminants from migrating west of Dunn Field (offsite). The role of this groundwater extraction system in the final remedy for Dunn Field is not known at this time.

Natural attenuation recently has become more widely recognized as a potentially significant mechanism for reducing groundwater contaminant levels, stabilizing plumes, and/or reducing plume size. If site conditions are appropriate, natural attenuation potentially could be as effective as groundwater extraction in reducing dissolved contaminant concentrations to cleanup goals at a much lower cost. Thus, it is important to evaluate the feasibility of using MNA to supplement or replace groundwater extraction.

If it cannot be demonstrated that the plume is stable and receptors are not at risk using natural attenuation only, it is beneficial to initiate an RPO evaluation if one has not been initiated previously. The RPO program provides for an annual review of site data to determine if the remediation system is making adequate progress toward meeting cleanup goals. If needed, the RPO program can be expanded to explore system optimization, new technologies, and/or regulatory opportunities to modify cleanup goals.

The RPO evaluation will facilitate determination of whether groundwater extraction is the most appropriate remedial approach for the site, based on the performance of the system to date, estimated CTC, and the remedial options that are currently available including emerging or newly developed technologies. If an alternative remedial approach

Figure 4.2 Decision Tree for Groundwater Extraction

is deemed more appropriate than groundwater extraction, then the sequence of events to follow may include an FS and/or pilot tests to further evaluate the alternative approach, preparation of design documents, and ultimately implementation of the alternative remedial action. It may be determined at this stage of the decision process that cleanup goals cannot be achieved in the aquifer, and a technical impracticability waiver of cleanup goals may be pursued, if appropriate. A new decision process would then be developed for the selected alternative remedial approach.

If groundwater extraction is determined to be the most appropriate technology, then operation of the system should continue; however, any modifications to reduce costs or improve efficiency that are recommended during the RPO evaluation should be implemented. Potential modifications may include adding or removing pumping wells from the system, altering flow rates, and/or modifying the treatment methodology. Also, it is possible that the demonstrated beneficial effects of natural attenuation could be used to support a reduction in pumping rates as a cost-effective modification.

During annual review of the monitoring data, contaminant levels in each of the sampled monitoring wells are reviewed and evaluated individually for compliance. Depending on the objective of the groundwater extraction system, compliance may consist of remediating the aquifer to specified cleanup goals, demonstrating effective containment, or both. If it can be demonstrated that aquifer cleanup has been achieved, then it is appropriate to seek regulatory approval to terminate operation of the extraction well(s) and initiate a post-termination monitoring program. In some cases it may be appropriate to discontinue operating a portion of the extraction system that is associated with a portion of a plume where concentrations have decreased below cleanup levels.

In many cases, asymptotic mass recovery is observed at the extraction wells before cleanup goals in the aquifer are reached. This is caused by a "tailing" effect. "Tailing" refers to the progressively slower rate of dissolved contaminant concentration decline observed with continued operation of a groundwater extraction system (USEPA, 1994). This effect can be observed in groundwater collected from monitoring wells as well as in the effluent from the extraction wells. Contaminant transport processes potentially responsible for "tailing" effects include: (1) diffusion of contaminants in low-permeability sediments, (2) hydrodynamic isolation within well fields, (3) desorption of contaminants from sediments, and (4) partitioning of relatively immiscible fluids into groundwater (USEPA, 1992 and 1994). If contaminant levels in the extraction well effluent decrease to near or below cleanup goals due to "tailing", while contaminant levels in the aquifer persist above cleanup goals (as evidenced by monitoring well data), there may be no additional benefit realized by continued pumping of the extraction well(s). At this point, the contaminant mass removed per unit volume of pumped water is very low. If asymptotic mass recovery is observed in any of the extraction wells, or if the extraction well effluent is below cleanup goals for three consecutive sampling events, then shutdown of the extraction well should be evaluated.

If continued pumping of an extraction well is critical to controlling plume migration in a sensitive area such that even short-term shutdown would pose an unacceptable risk to receptors, then the well should remain in operation even if asymptotic mass recovery has occurred or contaminant concentrations in the extraction well effluent are below cleanup goals. For example, if the contaminant plume is located near a receptor exposure point and pumping of a well creates a reversal of groundwater flow to protect receptors, then shutdown of the extraction well may pose an unacceptable risk. In this case, it would be appropriate to continue operation of the well until the risk has been reduced due to a decrease in the extent of the plume, a decrease in concentrations, or a change in the status of the exposure point (e.g., a drinking-water well taken off-line).

If continued pumping of an extraction well is not critical for containing a plume or otherwise mitigating risk, and asymptotic mass recovery has occurred or effluent concentrations are less than cleanup goals, then regulatory approval should be obtained to shut off the extraction well and sample it over time to assess potential rebound of contaminant concentrations. Contaminant "rebound" is believed to be caused by the same processes (described above) that are responsible for "tailing" effects (USEPA, 1992 and 1994).

If significant contaminant rebound occurs, then the appropriateness of groundwater extraction as the remedial action should be re-evaluated as part of the RPO annual review. If groundwater extraction continues to be viewed as the most appropriate technology, then pumping should continue until cleanup goals are achieved in the aquifer, contaminant rebound following extraction well shutdown is insignificant, or an alternative remedial approach is selected. It should be noted that "significant" rebound should be defined on a situation-specific basis in that there are no generally accepted guidelines.

If contaminant rebound is not significant, then regulatory approval for terminating operation of the extraction well should be sought. Under this scenario, aquifer cleanup goals may not have been met throughout the plume. If the primary objective of the pump-and-treat system is to remediate the aquifer to cleanup goals, then it should first be demonstrated that cleanup goals can be met within a reasonable time frame via natural attenuation. If this cannot be demonstrated, then it may be appropriate to pursue a technical impracticability waiver of cleanup goals and protect potential receptors via institutional controls (e.g., prevent groundwater extraction for drinking water purposes), select an alternative remedial technology, or continue groundwater extraction until a more effective alternative technology becomes available. If it can be demonstrated that aquifer cleanup goals can be met via natural attenuation, or if the primary objective of the extraction system is to contain the plume rather than remediate the aquifer, then it would be appropriate to seek regulatory approval to terminate extraction well operation if it can be demonstrated that the contaminant plume is stable, and/or there are no receptors that will be at risk when the extraction well is shut down.

Once the extraction well operation is terminated, post-termination monitoring to support long-term shutdown is initiated. Some examples of methods for conducting post-termination monitoring are provided by USEPA (1994). The post-termination monitoring plan should include evaluation of the potential for contaminant rebound and its significance. It is expected that some rebounding of COC concentrations will occur. When examining the significance of this, it is important to view the aquifer in a context larger than an individual monitoring point. For example, it may be appropriate to consider the average water quality of a portion of the aquifer rather than data from individual monitoring wells when evaluating whether or not concentrations below remedial goals are being maintained. For example, a water supply well installed in the affected aquifer may draw water from an area that is substantially larger than the non-compliance zone. When regulatory requirements for demonstrating protectiveness of human health and the environment have been met, then extraction and monitoring well sampling can be discontinued as part of the site closure process.

4.3 SOIL VAPOR EXTRACTION SYSTEM OPERATION AND EXIT STRATEGY

The SVE decision tree for DDMT (Figure 4.3) is based on the following assumptions:

- All contamination sources at the facility have been characterized;
- Initial pilot testing for SVE has been completed; and
- The SVE system has been designed to meet Tennessee's VOC emissions criteria (i.e., treatment of extracted vapors is not addressed in the decision tree).

Once these criteria have been met, the SVE system can be evaluated using the decision tree presented on Figure 4.3. The decision tree will help to identify the steps needed to optimize a system in order to reach soil cleanup goals, and to determine when a system can be shut down.

The first steps in the tree are to ensure that all source areas have been well-characterized and that permanent VMPs have been installed in appropriate soil intervals. In order to accomplish these tasks, vertical profiling tools such as Geoprobe's Membrane Interface Probe™ (MIP), or the SimulProbe™ manufactured by BESST, Inc. can be used to select optimal screen intervals for the VMPs. Screens should be placed in the most contaminated intervals and in the various lithologies present in the vadose zone. The profiling tools provide real-time data as a borehole is advanced. The MIP system allows the operator to continuously monitor soil contaminant concentrations while collecting continuous lithologic data. The SimulProbe™ was developed to measure soil gas quality and air permeability and to obtain soil samples as the borehole is being drilled. The SimulProbe™ can be used in conjunction with a conventional drilling rig (e.g., hollow-stem augering) or a cone penetrometer testing (CPT) rig.

During SVE system operation, extraction well effluent concentrations should be monitored until stable VOC concentrations are observed over a 3-month period. At this point, the SVE system should be shut down for 30-60 days to allow VOC vapors in the formation to equilibrate. VOC concentrations should then be monitored for rebound at all VWs and VMPs. If significant rebound of VOC concentrations occurs, vapor extraction flow rates and well screen intervals should be optimized for the soil volumes containing VOC residuals, and operations should continue until equilibrium VOC concentrations at all VMPs and VWs are below cleanup criteria. Examples of system optimization actions include:

- Adjustment of air flow rates to maximize mass removal per unit volume of air extracted and minimize vapor treatment costs;

Figure 4.3 Soil Vapor Extraction Decision Tree

- Turning off air flow to some wells to maximize flow to remaining contaminated intervals; and

- Installing one or more additional SVE wells with shorter screens to better target the remaining contaminated intervals.

The PneuLog™ tool can be used to assess the locations of remaining contamination "hotspots". PneuLog™ is a pneumatic well-logging device developed by PRAXIS Environmental Technologies, Inc. to measure soil vapor flow and contaminant profiles in existing VWs.

There are no generally accepted criteria for defining "significant" VOC rebound in soil vapors. Rebound to within 75% of initial concentrations is probably too low to be significant. However, rebound to within 25% to 50% of initial concentrations is potentially significant. Rebound to within 25% may be indicative of the presence of a continuing NAPL source. If VOC rebound concentrations are less than 25% of initial concentrations and below cleanup criteria, vapor-extraction should be directed away from clean areas toward remaining hot spots.

It is recommended that cleanup criteria be based on the relationship between average equilibrium soil gas concentrations in the soil column and the groundwater protection or risk-based soil standard for the site. Development of vadose-zone-soil cleanup levels is discussed in detail in Section 10.3.

SECTION 5

REVIEW OF NATURAL ATTENUATION DATA

Evaluating the potential for integrating MNA into the Depot's remediation strategy is an important part of the RPO process. The potential for biodegradation of CVOCs in groundwater at DDMT was evaluated in a study conducted by CH2M Hill in the spring of 2000. The purpose of this study was to assess whether MNA is a feasible remediation option for CVOCs dissolved in groundwater at the MI and Dunn Field. The results are reported in a “Natural Attenuation Technical Memorandum,” included as part of the final groundwater FS for the MI (CH2M Hill, 2000f). As part of the Phase II RPO evaluation, this document was reviewed for technical accuracy and completeness. Review comments are presented below, with appropriate page and subsection references to the technical memorandum.

5.1 GENERAL OVERVIEW

In general, CH2M Hill’s (2000f) conclusion that only limited and localized biologically-facilitated reductive dehalogenation of CVOCs is occurring at the MI and Dunn Field appears to be reasonable and correct. The strongest evidence supporting the occurrence of biological degradation is the presence of the PCE/TCE reductive dehalogenation daughter product *cis*-1,2-DCE within the plume area. The Dunn Field site exhibits a slightly greater potential for biodegradation of CAHs than the MI based on the results of the scoring process described by USEPA (1998) and Wiedemeier *et al*. (1999). However, the data supporting the occurrence of biodegradation do not outweigh the data that indicate that the CAH plumes at both sites can be classified as Type III, meaning that the groundwater system is characterized by inadequate concentrations of native and/or anthropogenic carbon, and concentrations of DO are greater than 1 milligram per liter (mg/L). These conditions are not supportive of reductive dehalogenation of PCE and TCE.

5.2 SPECIFIC COMMENTS

On page A-11, paragraph 2 of the "Natural Attenuation Technical Memorandum (CH2M Hill, 2000f), it is suggested that elevated TCE and *cis*-1,2-DCE concentrations potentially could be a result of reductive dehalogenation occurring in anaerobic, reducing microenvironments. Another possible explanation for elevated TCE concentrations is that TCE may have been disposed of at the site, rather than a daughter product of the reductive dehalogenation of PCE.

On page A-12, “Soluble Chloride Ion,” it is concluded that degradation of chlorinated solvents is occurring based on the interpretation of chloride data collected at the MI and

Dunn Field. Parsons ES finds the chloride data presented in the Natural Attenuation Technical Memorandum for both sites to be inconclusive with respect to providing evidence of chlorinated solvent degradation. Recently-obtained groundwater analytical data should be reviewed to clarify whether chloride concentrations really are significantly elevated above background concentrations in the CAH plume areas.

In the MI area, one chloride sample result (10 mg/L) is reported for a single background well (MW72). This information is insufficient for characterizing background chloride concentrations. Chloride concentrations at wells within the CVOC plumes (MW21, MW22, MW39, MW47, MW20, and MW62) ranged from 3 to 34 mg/L, with the highest chloride concentration occurring in the well that exhibits among the lowest PCE, TCE, and DCE concentrations (MW22). For chloride data to be used confidently as an indication of reductive dehalogenation, higher contaminant concentrations (in the mg/L range, similar to the chloride concentrations) probably would be required to noticeably elevate chloride concentrations above background levels.

In the Dunn Field area, the background chloride concentration is represented using data from only one well (MW46). Due to the lack of background chloride data, the observation that chloride concentrations at MW54 (located within the plume) and MW40 are elevated relative to background due to the occurrence of reductive dehalogenation cannot be substantiated. Recently-obtained groundwater analytical data should be reviewed to clarify whether chloride concentrations really are significantly elevated above background concentrations in the CAH plume areas.

On page A-16, "Sulfate/Sulfide," it is suggested that the potential for reductive dehalogenation is supported by low sulfate concentrations and the absence of sulfide in groundwater samples. Although the sulfate concentrations are sufficiently low that reductive dehalogenation of CVOCs should not be inhibited, they do not support the occurrence of this process. The presence, not absence, of sulfide is indicative of sulfate-reducing conditions that are conducive to reductive dehalogenation because sulfide is formed when sulfate is used as an electron acceptor in microbially mediated redox reactions.

On page A-16, "Methane," the highest methane concentration (0.04296 mg/L) was detected at the edge or outside of the CVOC plume area. Trace methane concentrations within the plume support the inference that strongly reducing conditions, which are supportive of reductive dehalogenation, are not present.

On page A-17, "Methane,", comparison of methane units in the text with those in Table 3 and Figures 21 and 22 indicates that the units of mg/L used in the text are incorrect; they should be µg/L.

On page A-18, "Ammonia," the word "reducing" should be changed to "redox" in the second sentence under the heading "Main Installation."

On page A-19, "Biochemical Oxygen Demand," the significance of the biological oxygen demand (BOD) data could be expanded upon. BOD is a measure of the total oxygen consumption by microorganisms during degradation of organic matter. Low

BOD is indicative of low concentrations of food-quality organic carbon, and is consistent with the low dissolved organic carbon (DOC) concentrations and oxidizing environments.

On pages A-19 and A-20, "Approximation of Degradation Rates", and in supporting Tables 8, 9, and 10, the method used to normalize the observed downgradient tracer concentration for estimation of the decay rate constant is not appropriate because the tracer (chloride-plus-chlorine) concentration increases along the flowpath. In the MI area the tracer concentration increases from 11.10 to 20.01 mg/L along the flowpath, and at Dunn Field the tracer concentration increases from 17.03 to 47.07 mg/L. The normalization method employed in Tables 8 through 10 requires that the tracer *decrease* along the flowpath primarily as a result of dispersion and/or dilution. Because the tracer is "conservative" it should not decrease or increase along the flowpath due to reactions or degradation. The method assumes that chlorine remains in balance along the flowpath (i.e., as the chloride ion is produced along the flowpath due to reductive dehalogenation, organic chlorine is depleted; thus the total chloride plus chlorine concentration remains the same). In theory, if a decrease in chloride-plus-chlorine concentration is observed along the selected flowpath, it is a result of dispersion or dilution.

Parsons ES re-estimated degradation rate constants using the method of Buscheck and Alcantar (1995). For this method the effects of dispersion are specified, thus data normalization is not required. The Buscheck and Alcantar (1995) method assumes that the plume is in steady-state equilibrium, and yields total destructive attenuation rates that account for chemical (abiotic) and biological (aerobic and anaerobic) decay. For an expanding plume, this first-order approximation can be viewed as an upper bound on the destructive attenuation rate.

Table 5.1 compares the results of the re-calculation and the results from Tables 8 through 10 (CH2M Hill, 2000f). The decay rate constants estimated using both methods resulted in similar values. This is because normalized concentration corrections (used in the tracer method) and the specified dispersivity (used in the Buscheck and Alcantar method) had only a small effect on the decay rate calculations (i.e., the calculations treated decay as the primary mechanism for contaminant reduction along the flowpath). Two PCE decay rates were estimated for the MI using the lower fraction of organic carbon (f_{oc}) value reported in Table 8 (0.0004) and the higher f_{oc} value (0.00286) used in Tables 9 and 10. It is unclear in the technical memorandum which of the two values better represents the f_{oc} in saturated soils at the MI. The f_{oc} value of 0.00286 compares well with the value of 0.002345 reported in the RI reports for the MI (CH2M Hill, 2000a) and Dunn Field (CH2M HILL, 2000b). Clarification of the f_{oc} content of the Fluvial aquifer is required to better estimate degradation rate constants.

On page A-20, 2nd bullet, "Summary and Conclusions for the MI," wells MW24 and MW34 are not source area wells as stated. Based on the data presented in the report, these wells are located just outside the plume boundary. Nor is well MW20 a "downgradient" well, as stated. This well also is located near the periphery or outside of (crossgradient from) the CAH plume. This suggests that the chloride concentrations detected in this well may represent background conditions rather than an indication of biodegradation.

On page A-20, 4th bullet, "Summary and Conclusions for the MI," sulfate concentrations less than 20 mg/L do not support reductive dehalogenation as stated; rather, one can conclude that this process should not be significantly inhibited by the available sulfate (see previous comment for page A-16, "Sulfate/Sulfide").

TABLE 5.1
SUMMARY OF ESTIMATED DEGRADATION RATE CONSTANTS AND HALF-LIVES
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

	Degradation Rate ($year^{-1}$)		Half-Life (year)	
	Tracer Method (Tables 7,8,9)[a/]	**Buscheck and Alcantar Method**[b/]	**Tracer Method (Tables 7,8,9)**[a/]	**Buscheck and Alcantar Method**[b/]
Main Installation:				
PCE	0.08605[c/] - 0.2151[c/]	0.090[c/], 0.026[d/]	8.0 [c/] - 3.2[c/]	7.7[c/], 28[d/]
TCE	0.06219 - 0.1798	0.053	11 - 3.9	13
Dunn Field:				
PCE	not estimated	0.022[d/]	not estimated	32[d/]
TCE	0.0930-0.199	0.16	7.5 - 3.5	4.3

a/ CH2M Hill (2000f).
b/ Buscheck and Alcantar (1995) method calculations are provided in Appendix B.
c/ Calculation assumes a PCE retardation factor of 1.63 based on 0.0004 organic carbon.
d/ Calculation assumes a PCE retardation factor of 5.5 based on 0.00286 organic carbon.

On page A-22, 2nd bullet, "Summary and Conclusions for Dunn Field," it is suggested that the occurrence of elevated ethene/ethane concentrations is a direct indication that PCE/TCE are being reductively dechlorinated. The highest detected concentration of ethane/ethene was 0.2 μg/L, which indicates that very limited production of these compounds is occurring. The degree to which PCE and TCE are being reductively transformed is better indicated by *cis*-1,2-DCE concentrations.

On page A-22, 4th bullet, "Summary and Conclusions for Dunn Field," it is suggested that nitrate concentrations in the vicinity of well MW31 support reductive dehalogenation. Well MW31 is at the periphery of the plume; therefore, low nitrate concentrations in this area are not very significant. In addition, the detected concentration (0.9 mg/L) is virtually equivalent to the threshold value (1.0 mg/L) given in the scoring table, so the conclusion that it supports reductive dehalogenation is tenuous at best.

On page A-22, 5th bullet, "Summary and Conclusions for Dunn Field," it is suggested that because sulfate concentrations in several of the source area wells and the background well are less than 20 mg/L that reductive dehalogenation is supported. Rather, the fact that sulfate concentrations in the source wells are similar to background concentrations indicates that sulfate is not being used as an electron acceptor in microbially mediated redox reactions because the groundwater system is too oxidizing. The lack of depleted sulfate concentrations relative to background supports the conclusion that the groundwater system is not sufficiently reducing to promote reductive dehalogenation (see comment on page A-20, 4th bullet).

On page A-22, 6th bullet, "Summary and Conclusions for Dunn Field," it appears that the units quoted are incorrect (see comment for page A-17, "Methane"). None of the wells sampled contained methane concentrations greater than the threshold of 0.5 mg/L.

5.3 CONCLUSIONS ON EFFECTIVENESS OF MONITORED NATURAL ATTENUATION AS A REMEDIAL ALTERNATIVE:

CH2M Hill (2001a) concludes that between 15 and 50 years would be required to reduce MI plume concentrations to MCLs based on the calculated decay rates presented in the groundwater FS (CH2M Hill, 2000f). Based on the first-order biodegradation rates computed using the method of Buscheck and Alcantar (1995) (Table 5.1), the maximum PCE concentration detected on the MI in March 2000 (78 μg/L in well MW21) would decrease to the MCL of 5 μg/L in 34 to 113 years (years 2035 and 2114, respectively). The discrepancy between these remediation time frames is caused by the use of differing f_{oc} concentrations in the decay rate calculations; this discrepancy highlights the utility and necessity of clarifying the f_{oc} content of the Fluvial aquifer.

The estimated time frame for the maximum TCE concentration detected in the Dunn Field plume in March 2000 (1,200 μg/L at well MW70) to decrease to the MCL of 5 μg/L under the influence of natural attenuation is 38 years (year 2039) based on a decay rate of 0.16 yr^{-1} (Table 5.1). The remedial time frame estimates for TCE and PCE assume that the contaminant source has been effectively removed, which is not the case for Dunn Field, and may not be true for the MI. Therefore, reliance on MNA alone would potentially require a commitment to many years of groundwater monitoring if groundwater cleanup goals remain equivalent to MCLs. The natural attenuation evaluations performed by CH2M Hill (2000f) and as part of this RPO Phase II evaluation support the inference that natural bioremediation of dissolved TCE and PCE in Depot groundwater is limited, and confirm the need for source removal to accelerate attainment of groundwater cleanup goals.

SECTION 6

ASSESSMENT OF ENHANCED BIOREMEDIATION

As described by CH2M Hill (2001a), the selected remedial alternative for MI groundwater includes enhanced *in situ* bioremediation of CVOCs. The use of HRC® is assumed for costing purposes in the draft RPO for the MI. This section focuses on the evaluation of enhanced *in situ* bioremediation through organic substrate addition to achieve both source control and mitigation of CVOC plumes at DDMT. The evaluation focuses on the use of vegetable oil as an organic substrate as a potential alternative to HRC®. The objective of this evaluation is to determine the feasibility of enhanced *in situ* bioremediation (i.e., stimulating biodegradation) to provide an effective solution to groundwater contamination that also is cost-effective, will reduce the overall cleanup time, and will minimize the risk of contaminant rebound in the dissolved plumes. The enhanced bioremediation technologies discussed in this section are intended to be used in conjunction with MNA.

6.1 DESCRIPTION OF ORGANIC SUBSTRATE ADDITION

Reductive dehalogenation is a known mechanism for the biodegradation of many chlorinated solvents. Previously developed laboratory and field data have shown that reductive dehalogenation occurs under reducing conditions, where an electron donor is utilized as the main energy source for microbial metabolism. Because CAH compounds are used as electron acceptors, there must be an appropriate source of carbon for microbial growth in order for reductive dehalogenation of CAHs to occur. Hydrocarbon fuels, landfill leachate, and natural carbon sources are examples of organic substrates that can act as electron donors. Substrates utilized for enhanced bioremediation include both solid and liquid forms and range from readily soluble (e.g., lactate, acetate, methanol, molasses, or glucose) to fairly insoluble (e.g., vegetable oils, sawdust, bark mulch, polymers, and hydrogen-releasing compound [HRC®]) forms.

Relative to pump-and-treat methods, *in situ* groundwater cleanup methods reduce costs, limit infrastructure disruption, and minimize waste streams requiring treatment and disposal. However, experience has proven that an obstacle to successful competitive development of bioremediation processes for CVOCs is often the cost-effectiveness of nutrient or substrate addition methods. For example, although oxidative cometabolism of CVOCs during biodegradation of another carbon substrate has been shown to be an effective means of bioremediation, the costs of nutrient or substrate addition have severely limited its commercial acceptance. Reductive dehalogenation appears to require less substrate mass, and should therefore be more cost-effective.

The most common enhanced-bioremediation approach utilized to date has been addition of a carbon source dissolved in groundwater. Typically, these soluble sources are transported with bulk groundwater movement and move as a solute front. Movement of the introduced carbon source away from the contaminant source area does not provide long-term (e.g., 5 or more years) control of mass flux from the contaminant source area. Soluble carbon sources create anaerobic zones suitable for reductive dechlorination soon after their introduction into the saturated zone, but are quickly degraded. Therefore, soluble sources generally require frequent and costly replacement because their effect is short-lived.

Slow-release carbon sources include HRC® and food-grade vegetable oil. Although HRC® is a slow-release source, it also moves as a solute front and does not provide long-term control of mass flux from a contaminant source area. In addition, the cost of HRC® per unit of carbon mass is generally one to two orders of magnitude higher than vegetable oils. Other approaches involving the placement of solid materials that release carbon (e.g., bark mulch) are promising, but the cost of carbon placement is high.

6.2 VEGETABLE OIL INJECTION

Vegetable oil injection (the VegOil process) is an innovative, cost-effective method of carbon addition that promotes the redox and electron-donor conditions necessary to promote *in situ* microbial dehalogenation of solvents in groundwater.

6.2.1 Advantages

- Vegetable oil is an inexpensive ($0.20 to $0.50 per pound), innocuous, food-grade carbon source that is not regulated as a contaminant by the USEPA.

- Vegetable oil can be injected directly into an affected aquifer via conventional wells in sufficient volume to ensure wide distribution throughout a contaminant plume. Injection permits must be obtained from the Memphis and Shelby Counties Water Quality Control Board and the TDEC. Injection permits were obtained for the Naval Support Activity Mid-South Site described in Section 6.4.

- Vegetable oil has an aqueous solubility that ranges from approximately 100 to 1,000 mg/L, and does not quickly dissolve into groundwater, thus serving as a slow-release carbon source.

- Vegetable oil tends to adsorb to soil particles, and therefore is less mobile than HRC® and does not move as a solute front.

- A single injection can potentially provide sufficient carbon to drive reductive dehalogenation for several years. This significantly lowers operation and maintenance (O&M) costs compared to aqueous-phase carbon injection, and allows injection of a much greater quantity of carbon than does solid-phase carbon emplacement.

- These properties allow a single introduction of a long-term carbon source located near the contaminant source area that will stay within the zone in which it was originally placed.

- The octanol/water partition coefficient for TCE is approximately 300, indicating that TCE has a much stronger affinity for entering an organic phase than to dissolve into groundwater. Dissolved chlorinated solvents will thus partition into the vegetable oil, thereby reducing the aqueous-phase contaminant concentrations in the source area until steady-state conditions are reached. Therefore, the process is effective both for accelerating biodegradation in a contaminant source zone, and for limiting downgradient contaminant migration.

6.2.2 Disadvantages

While creation of an anaerobic zone to facilitate reductive dehalogenation has great potential to control the flux of CVOCs and facilitate overall remediation of the dissolved plume, there also are potential deleterious side-effects associated with any carbon addition technique. These side-effects may include:

1. VC could be generated at a rate greater than the rate of its anaerobic reduction or aerobic oxidation, thus potentially leading to accumulation of VC;
2. Trace metals potentially incorporated in (and coprecipitated from) iron or manganese oxides could be solubilized under reducing conditions.
3. Methane could be generated by methanogenic bacteria degrading the introduced carbon under anaerobic conditions.

These side-effects have not been studied in detail, as the VegOil technology has only recently been applied. However, evaluation of the properties of VC, trace metals, and methane can alleviate most concerns regarding their potential effects. These properties include:

- VC can be degraded both aerobically and anaerobically. Reductive dehalogenation can completely dehalogenate VC (although this can be a relatively slow process). VC is also readily oxidized and should quickly be degraded in an aerobic environment such as exists in groundwater beneath DDMT. Also, VC can easily be stripped from aqueous streams.

- There is sufficient iron in groundwater at the MI and Dunn Field to suspect that iron oxide exists; however, the quantity of metals coprecipitated with iron, if any, is difficult to assess. Trace metals potentially released from an anaerobic treatment zone should oxidize and precipitate in a downgradient aerobic zone, limiting their mobility. In addition, increased metals concentrations in groundwater may not be a concern because the Fluvial aquifer is not used as a water supply source.

- Methane produced in the reaction zone will be in the dissolved state, which does not present a concern. If concentrations reach a solubility limit, methane could potentially be transported to the vadose zone as a gas, where it could become an explosion hazard. However, based on evaluation of chlorinated solvent sites where reductive dehalogenation is known to occur, it is reasonable to assume that the methane concentrations would not be high enough to be a cause for concern.

6.3 APPLICABILITY OF VEGETABLE OIL INJECTION AT DDMT

The applicability of VegOil injection was evaluated for both the MI and Dunn Field. At both sites the CVOC plumes can be classified as Type III (i.e., plumes that have a soluble mass of PCE and TCE, but do not contain a significant carbon source (USEPA, 1998). Although some degradation of PCE and TCE is occurring at both sites, as evidenced by the presence of the reductive dehalogenation daughter product *cis*-1,2-DCE in a number of wells, the ultimate degradation of PCE and TCE to innocuous daughter products is limited by the low organic carbon content in the substrate and the relatively oxidizing redox conditions.

Carbon addition is implemented in a manner that produces a permeable reactive zone in which the groundwater becomes anaerobic and reductive dehalogenation can occur. For effective reductive dehalogenation of PCE/TCE to occur, the environmental conditions must be altered from the general aerobic conditions of the aquifer to an anaerobic condition. Once anaerobic conditions prevail, reductive dehalogenation of PCE/TCE to *cis*-1,2 DCE to VC to ethene and other innocuous byproducts can occur.

While the rate of reductive dehalogenation of VC may be slow relative to that of PCE, TCE, and DCE under anaerobic conditions, VC generally degrades rapidly under aerobic conditions. To completely degrade VC to CO_2, water, and chloride, it is beneficial to have an aerobic zone downgradient of the reductive zone. The important consideration is that PCE and TCE do not readily degrade under aerobic conditions such as those found at the MI and Dunn Field. Once PCE and TCE are converted to DCE and VC under induced anaerobic conditions, the complete degradation of daughter products to CO_2, water, and chloride can be achieved under a combination of anaerobic (reductive dehalogenation of PCE, TCE, DCE, and VC) and aerobic (oxidation of DCE and VC) conditions.

Geochemical parameters measured at the site indicate that the amount of available carbon is insufficient to create a reducing (i.e., sulfate-reducing or methanogenic) environment that is supportive of significant reductive dehalogenation. The addition of vegetable oil is designed to overcome this organic substrate deficiency and to induce the reducing environment required for significant reductive dehalogenation of chlorinated solvents.

To gain a better understanding of how enhanced *in situ* bioremediation might effect the time required to reach cleanup goals, decay rates and half-lives estimated for the natural conditions at the MI and Dunn Field (Table 5.1) were compared to literature values reported by Parsons ES (1999b) for conditions typical of enhanced bioremediation. The literature values represent an average for six sites where reductive dehalogenation is known to occur under Type I conditions (anaerobic environment with substantial anthropogenic carbon). The decay rates and half-lives used in the comparison are summarized in Table 6.1. The method of estimating the decay rates for the MI and Dunn Field under natural conditions is presented in Section 5.2 and Appendix B.

TABLE 6.1
SUMMARY OF FIRST ORDER DECAY RATES
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

	Decay Rate (year^{-1})	Half-Life (year)	C_o (μg/L)[a]	T[b] (year)
Main Installation Natural Conditions [c]				
PCE	0.090	7.7	200, 78 [d]	42, 30
	0.026	28	200, 78	147, 109
TCE	0.053	13	-- [e]	--
Dunn Field Natural Conditions [c]				
PCE	0.022	32	--	--
TCE	0.16	4.3	1,200	34
Enhanced Bioremediation [f]				
PCE	0.231	3	200, 78	17, 13
TCE	0.231	3	1,200	24

[a] Co = initial concentration; μg/L = micrograms per liter.
[b] t = time estimated for C_o to decay to cleanup levels of 5 μg/L PCE and TCE.
[c] Rates and half-lives from Table 5.1, computed using the Buscheck and Alcantar (1995) method.
[d] Maximum concentration detected off-site (200 μg/L) and onsite (78 μg/L) in March 2000.
[e] -- Not evaluated
[f] Average rates and half-lives from six Type I sites reported in Parsons ES (1999b).

The times required for maximum-detected concentrations of PCE and TCE to decay to the cleanup goal of 5 μg/L at the MI and Dunn Field, respectively, were estimated for both natural and enhanced bioremediation conditions using the decay rates presented in Table 6.1. These constituents were selected because PCE is the predominant CAH in groundwater at the MI and TCE is the predominant CAH in groundwater at Dunn Field. These estimates were made using the first-order decay equation $C=C_oe^{-kt}$, where C is the cleanup level concentration, C_o is the initial concentration (based on maximum concentrations detected in March 2000), k is the decay rate estimated for a particular site and contaminant, and t is the time estimated for the initial concentration to decay to the cleanup level. The values used for C_o and the estimated t values are presented in Table 6.1. For these estimates it was assumed that the contaminant source has been removed.

Under natural conditions, the estimated time required to reach cleanup goals for PCE at the MI and TCE at Dunn Field is at least 30 years (Section 5.3). Assuming that the enhanced biodegradation decay rates in Table 6.1 are representative of site conditions under an enhanced-bioremediation scenario, then the time required to reach the PCE cleanup goal at the MI is reduced to 13 to 17 years, and the time required to reach the TCE cleanup goal at Dunn Field is reduced to approximately 24 years. These time frames are longer than the 10 years estimated by CH2M Hill (2001a). It should be noted that these results could be in error by at least an order of magnitude considering the uncertainty inherent in estimating decay rates. A more representative evaluation of the effectiveness of enhanced bioremediation could be made by conducting an enhanced-bioremediation pilot test at DDMT.

6.4 RESULTS FROM OTHER SITES AND REGULATORY ACCEPTANCE

The VegOil approach has been applied at several sites across the United States including Travis AFB, California; two Army sites at Defense Depot Hill, Utah (DDHU); Naval Support Activity Mid-South in Memphis, Tennessee; and Cape Canaveral Air Station in Florida. At DDHU, a pilot test was recently completed with results supporting significant reduction of CVOC concentrations within the plume. A full-scale system was installed in July 2000, and, based on the first round of post-injection sampling, concentrations of TCE, *cis*-1,2-DCE, and VC have decreased below the laboratory detection limit, and methane generation is occurring. At the Mid-South facility in Memphis, the results of the first round of post-injection sampling conducted 3 months after injection indicate that the groundwater system in the immediate vicinity of the injection wells is becoming more reducing, methane is being produced, and reductive dehalogenation is being stimulated; additional monitoring will be required to draw conclusions on the degree of enhanced biodegradation that is occurring. A 6-month sampling event is scheduled for February 2000. The results of this test will be particularly helpful in assessing the VegOil approach for DDMT because the site contaminants (CVOCs) and hydrogeology at Mid-South are similar to that of DDMT. The targeted (Fluvial) aquifer at Mid-South is comprised of sand and gravel. The oil injection wells are screened between 45 and 85 feet bgs. However, the groundwater at Mid-South is generally less aerobic than at DDMT, with initial DO concentrations ranging from 0.3 to 3.4 mg/L. Additional applications of VegOil have been implemented at Edwards AFB, California and Dover AFB, Delaware.

Vegetable oil injection and subsequent creation of a reactive zone has been accepted by the USEPA (Travis AFB, California; DDHU; Naval Support Activity Mid-South, Tennessee) and the State of California Regional Water Quality Control Board (Travis AFB and Edwards AFB, California). Other state regulatory agencies approving application of oil injection include the Utah Department of Environmental Quality, TDEQ, and the Florida Department of Environmental Protection.

6.5 IMPLEMENTABILITY AND COST

Although groundwater conditions at both the MI and Dunn Field sites are suitable for enhanced bioremediation, there are some logistical and implementability issues to be considered for both sites. At Dunn Field, the CVOC plumes have already migrated off-site. During enhanced bioremediation, production of VC will occur. It is not feasible to accurately predict how much VC would be produced and how long it would persist before it is aerobically degraded to innocuous byproducts. However, VC has the greatest tendency to undergo oxidation of any of the chlorinated ethenes, and rapid microbial degradation of VC has been observed in laboratory cultures and aquifer samples under aerobic conditions (Bradley, 2000). Despite the strong possibility that VC would either be captured by the existing groundwater extraction system or would degrade prior to reaching a receptor exposure point, the fact that a relatively toxic compound would be produced near the Depot boundary or off-site may be viewed as unacceptable by regulators or the local community. Assuming that any contaminant source present in the vadose zone will be removed via SVE, it would be useful to first assess the effectiveness of the existing groundwater extraction system at Dunn Field for at least two years prior to evaluating alternative or supplemental remedies for groundwater such as enhanced bioremediation. The effectiveness evaluation should consist of monitoring dissolved contaminant concentrations in monitoring wells and extraction well effluent.

Of the two sites, the MI would be the most suitable for an enhanced bioremediation pilot study from an implementability standpoint. The southwestern CAH plume appears to be migrating onto the installation from off-site. The plume does not appear to pose a threat to downgradient receptors (see Section 7), and currently there are no engineered remedial systems in place. The use of enhanced bioremediation could prove to be a cost-effective engineered remedial measure for MI groundwater. If it can be concluded from a pilot study that enhanced bioremediation is effective at the MI, then this technology could be used to reduce CVOC concentrations within the plume, and if the source is later identified and removed, then enhanced bioremediation potentially could be used to remediate groundwater to cleanup goals.

The estimated cost to conduct a pilot study would be on the order of $220,000 (Appendix E). Pilot testing would include the following tasks:

1. Preparation of a work plan;

2. Field Activities;

    - Drilling/soil sampling/injection and monitoring/well installation and development/surveying,

    - Baseline well sampling,

    - Aquifer (slug) testing,

    - Oil injection, and

    - Performance monitoring; and

3. Data analysis and reporting.

The pilot test should consist of at least two to four VegOil injection wells and four to eight monitoring wells. For this cost estimate, it was assumed that two injection wells and four monitoring wells would be installed, and that a total of five rounds of groundwater sampling would be conducted, including the baseline sampling event. Also, it was assumed that Parsons ES-Denver would conduct the field work. Costs to conduct the pilot study could be reduced if a local firm were to conduct the four performance monitoring rounds.

A more detailed cost proposal to conduct a VegOil pilot test can be prepared if further investigation of this technology is desired. The cost proposal would include a work breakdown structure and would provide specific details regarding the field work (e.g., well locations and depths, chemical analyses, etc.), as well as the assumptions associated with the costs.

SECTION 7

EVALUATION OF MAIN INSTALLATION PLUME STABILITY

7.1 RESULTS OF BIOCHLOR MODELING

7.1.1 Model Description

The BIOCHLOR software program (Aziz *et al*., 1999) is a screening-level model that simulates the natural attenuation of aqueous-phase chlorinated solvents. It is currently under review for clearance and release as a USEPA document. BIOCHLOR is based on a semi-analytical solution of the Domenico (1987) reactive transport model. Transport of dissolved chlorinated solvents under the influence of one-dimensional (1-D) advection, three-dimensional (3-D) dispersion, linear adsorption, and first-order decay can be modeled using the BIOCHLOR code.

BIOCHLOR is appropriate for:

1. Assessment of plume migration rate, distance, and spatial distribution over time with respect to potential downgradient receptors; and

2. Use as a screening tool for determining whether remediation by natural attenuation (RNA) is a feasible option for chlorinated plumes (Aziz *et al*., 1999).

For an analytical model, hydrogeologic conditions must satisfy the uniformity conditions within the model domain. Although unable to simulate non-laminar groundwater flow, BIOCHLOR, when properly constrained to a uniform flow field, may be reliably used to ascertain the fate and transport of a solvent plume (American Society for Testing and Materials [ASTM], 1995; Aziz *et al*., 1999; Domenico and Schwartz, 1990).

7.1.2 Modeling Objectives

Analytical models are useful in cases where a lack of data would make implementation of a more sophisticated numerical model inappropriate. Analytical models are generally considered to be effective primary screening tools for natural attenuation decision-tree analysis (USEPA, 1998). The primary modeling objective for DDMT was to assess whether the PCE and TCE plumes near the southwestern corner of the MI could potentially migrate to monitoring well MW34 at concentrations of concern. Previous studies have indicated the potential presence of a "window" or gap in the clay layer underlying the Fluvial aquifer in the vicinity of MW34 that could allow migration of groundwater and dissolved contaminants into deeper zones. The BIOCHLOR modeling software was used to estimate the maximum future concentrations of PCE and TCE at MW34 given constant-strength contaminant sources. Given the alternate conceptual groundwater flow model for the Fluvial aquifer presented in the MI ROD (CH2M Hill, 2001a), a secondary modeling objective was to assess the potential for chlorinated solvents dissolved in groundwater near the southwestern corner of the MI to migrate in a southeasterly direction toward installation boundary well MW24 from the vicinity of MW39 (Figure 2.8).

The model simulation results described in this section can be used to evaluate the degree to which dissolved CVOC contamination near the southwestern corner of the MI poses a risk to potential off-Depot receptors. The results also can be used to help finalize the scope (i.e., the numbers and locations of nutrient/chemical injection wells) of the planned enhanced bioremediation effort (see Section 5).

7.1.3 Previous Modeling Efforts

A previous modeling effort using BIOSCREEN (Newell *et al*., 1996) was performed during the FS (CH2M Hill, 2000f). Unlike BIOCHLOR, BIOSCREEN is not capable of simulating sequential reductive solvent decay; however, an assumed first-order decay coefficient equal to 20 percent of the calculated decay coefficients for the Fluvial aquifer was used to model TCE migration under the influence of biodegradation. The fate and transport of a hypothetical release of PCE and TCE into the Memphis Aquifer below the potential gap in the overlying confining unit near MW34 was examined. Using a constant-strength source input of 10 µg/L for both PCE and TCE, maximum dissolved concentrations of both PCE and TCE of 4 µg/L were predicted to reach the nearest Allen Well Field well (7,000 feet away) 45 years (PCE) and 90 years (TCE) after the initial release. The BIOSCREEN model results are considered to be conservative for the following reasons:

- The assumption of a constant source;
- The assumption that the introduced concentrations of PCE and TCE (0.010 mg/L) are not attenuated as these compounds are transported from the Fluvial aquifer to the Memphis Sand aquifer;
- The low magnitudes of the first-order decay coefficients used in the model; and
- The fact that volatilization within the Allen Well Field’s pumping and manifold systems is ignored.

Therefore, the BIOSCREEN results provide a measure of confidence that the Allen Well Field would not be adversely impacted given the modeled PCE/TCE concentrations in the source area.

7.1.4 Conceptual Model

Available groundwater quality data indicate that the PCE plume(s) originate near the southwestern corner of the MI near MW47 and PZ04 (Figure 7.1). PCE concentrations

Figure 7.l PCE Concentrations in Main Installation Groundwater March 2000

detected at well MW47 in March 2000 were assumed to be representative of source area concentrations. Monitoring well MW34 at the northern boundary of the MI was chosen as a point of compliance for the model simulations due to its proximity to the potential gap in the clay aquitard that could indicate an hydraulic connection between the Fluvial aquifer and deeper, confined aquifer underlying the Fluvial aquifer (Graham and Parks, 1986).

Information presented in the RI report for the MI (CH2M Hill, 2000a) suggests the presence of a bi-directional flow field along the modeled flowpath between the inferred source area and well MW34. As shown on Figures 2.7 and 7.2, groundwater appears to migrate to the northeast from the vicinity of PZ04 and MW47. Figure 2.7 suggests that groundwater in the vicinity of MW39 migrates in a northwesterly direction toward MW34, presumably channeled by a topographic depression in the underlying clay unit (Figure 2.6). In order to use BIOCHLOR to simulate this bi-directional flowpath, it was necessary to model the flow field as two separate model domains, hereafter referred to as Flowpaths F1 and F2 (Figure 7.3). This approach satisfies the linear, uniform-flow-field requirements of the BIOCHLOR model. Flowpath F1 begins near PZ04 and terminates at MW39 with a length of approximately 2,700 feet. Flowpath F2 begins at MW39 and ends at MW34, spanning a length of approximately 2,250 feet.

The distribution of TCE in groundwater beneath the MI in March 2000 is shown on Figure 7.4. Comparison of Figures 7.1 and 7.4 suggests that the PCE and TCE plumes are migrating along different flowpaths. Unlike the PCE plume, the TCE migration direction indicated on Figure 7.4 does not correspond to the groundwater flow direction inferred from potentiometric surface maps for the Fluvial aquifer constructed using November 1998 and March 2000 data (Figures 2.7 and 7.2). The similarity of TCE concentrations measured at MW21 (39 μg/L) and MW62 (32 μg/L) suggests that at least one additional source of TCE may exist between these two wells or near MW62 (i.e., the detected TCE concentration at MW62 may not result entirely from migration of TCE from the vicinity of MW21). Therefore, the TCE migration pathway simulated using BIOCHLOR extends from MW62 to MW34 (Figure 7.3), hereafter referred to as Flowpath F3. The upgradient portion of the plume (i.e., southwest of MW62) was not simulated.

A revised water table map that includes only data obtained from wells screened in the Fluvial aquifer is presented in the MI ROD (CH2M Hill, 2001a) (Figure 7.5). Based on these data, groundwater is inferred to migrate from the vicinity of MW39 in a southeasterly direction toward installation boundary well MW24. To date, PCE has not been detected in groundwater from MW24. The BIOCHLOR model was used to predict whether the PCE detected at MW39 could potentially migrate to MW24. The simulated migration pathway between MW39 and MW24 is referred to as Flowpath F4 (Figure 7.3).

7.1.5 Model Input

Input into the model consists of the advective groundwater velocity (V_x), dispersivity in three dimensions (α_x, α_y, α_z), adsorption, first-order biodegradation rates (or half-lives, $t_{1/2}$) for each COC, desired simulation time, model dimensions (length, width), and source area size and strength. Model input used in the predictive simulations for Flowpaths F1,

Figure 7.2 Potentiometric Surface Map of the Fluvial Aquifer March 21, 2000

Figure 7.3 Simulated Flowpaths for PCE and TCE

Figure 7.4 TCE Concentrations in Main Installation Groundwater March 2000

Figure 7.5 Water Table Elevations in Fluvial Deposits April 2000

F2, F3, and F4 are presented in Table 7.1. Simulation-specific model input summaries are provided in Appendix C.

7.1.5.1 Advection Rate

Advection refers to the process by which solutes are transported by the bulk motion of the flowing groundwater. It is a function of the horizontal groundwater velocity (V_x), which is based on the horizontal hydraulic conductivity (K_x), effective porosity (n_e), and horizontal hydraulic gradient (I_x). The geometric mean of the MI K_x values presented by CH2M Hill (2000a) (2.2E-03 cm/sec) was used for each simulation. The effective porosity of the Fluvial aquifer was estimated to be 0.30 using published ranges (Fetter, 1994). The average I_x between MW47 and MW39 was computed to be 0.0037 ft/ft using March 2000 water-level data (CH2M Hill, 2000f). The resultant V_x of 28.1 feet per year (ft/yr) was then used for all three simulated flowpaths. Use of the same V_x for each simulated flowpath was based primarily on the fact that MW34 may not be screened within the same flow field as MW39, MW47, and MW62. A review of monitoring well construction data indicates that MW34 is screened approximately 20 feet deeper than the other three wells. If MW34 is screened within a hydraulically dissimilar system, the water table surface as shown on Figure 2.7 may not adequately reflect the horizontal hydraulic gradients along Flowpaths F2 and F3 (Figure 7.3). Installation of nested piezometers at this location would be required to confirm this hypothesis. Although the primary simulations were run using an average hydraulic gradient of 0.0037 ft/ft as described above, additional, worst-case simulations for Flowpaths F2 and F3 were run using the steeper hydraulic gradients computed using the measured water level elevation at MW34. All of the simulation results are described in Section 7.1.7.

7.1.5.2 Dispersion

Dispersion refers to the process of solute mass spreading longitudinally (in direction of groundwater flow, [α_x]), transversely (perpendicular to groundwater flow, [α_y]) and vertically downwards (α_z) through interaction with the aquifer porous media, causing mechanical dispersion and chemical diffusion. Accurate measurement of this parameter is difficult in the field, and is further complicated by scale effects. However, simple estimation techniques based on the plume length or distance to a measurement point are available (Aziz *et al.,* 1999). The longitudinal dispersivity for each flowpath was selected to equal 10 percent of the flowpath length. Therefore, the magnitude of α_x varied for each flowpath. The transverse dispersivity was selected to be 10 percent of the longitudinal dispersivity at each measurement point within the model domain (Gelhar *et al*., 1992). The vertical dispersivity was set at the very low model default value of 1.0E-99, which indicates no vertical dispersion, consistent with model assumptions of a fully-penetrating source thickness (Aziz *et al.*, 1999).

7.1.5.3 Adsorption

Adsorption is a hydrophilic electrostatic surface interaction between a polar solute molecule and a negatively charged soil surface. This interaction tends to retard or decrease the rate at which the solvent plume is advectively transported within the aquifer. Aquifer parameters used to compute retardation coefficients (R values) include chemical-

TABLE 7.1
BIOCHLOR MODEL INPUT DATA
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Input Parameter	Flowpath F1	Flowpath F2	Flowpath F3	Flowpath F4	Data Source
Chemical evaluated	PCE	PCE	TCE	PCE	CH2M Hill (1997, 2000a) remedial investigations.
Hydraulic Conductivity, K_x (cm/s)	2.2E-03	2.2E-03	2.2E-03	2.2E-03	Geometric mean for the Main Installation (CH2M Hill, 2000e)
Hydraulic Gradient, I_x (-)	0.0037	0.0037, 0.0155	0.0037, 0.0344	0.0007	Calculated from Figures 2.7, 7.2, and 7.5
Effective Porosity, n_e (-)	0.30	0.30	0.30	0.30	Estimated using typical value for sand (Fetter, 1994; CH2M Hill 2000a).
Longitudinal Dispersivity, α_x (ft)	270	225	116	180	One-tenth of the length of each simulated flowpath
Transverse Dispersivity/Longitudinal Dispersivity ratio, α_y/α_x (-)	0.10	0.10	0.10	0.10	One-tenth of the longitudinal dispersivity (Gelhar *et al.*, 1992)
Vertical Dispersivity/Longitudinal Dispersivity ratio, α_z/α_x (-)	1E-099	1E-099	1E-099	1E-099	Model default value, fully penetrating source term (Aziz et al., 2000).
Soil Bulk Density, ρ_b (kg/L)	1.67	1.67	1.67	1.67	CH2M Hill (2000a)
Fraction organic carbon, f_{oc} (-)	0.0029, 0.0004	0.0029, 000.4	0.0029	0.0029, 0.0004	CH2M Hill (2000a, 2000b, 2000f)
Organic Carbon Partition Coefficient, K_{oc} (L/kg)	284	284	118.5	284	CH2M Hill (2000a)
Retardation Factor, R_f (-)	5.5, 1.63	5.5, 1.63	2.9	5.5, 1.63	Table 5.1
1^{st} Order Decay Coefficient, λ (1/yr)	0.025, 0.090	0.025, 0.090	0.053	0.025, 0.090	Table 5.1
Simulation Time, t (yrs)	100	100	100	100	Arbitrarily-selected simulation length

TABLE 7.1 (Continued)
BIOCHLOR MODEL INPUT DATA
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Input Parameter	**Flowpath F1**	**Flowpath F2**	**Flowpath F3**	**Flowpath F4**	**Data Source**
Modeled Area Width, y (ft)	1,500	1,000	1,000	500	Estimated from concentration data and interpolation scheme (CH2M Hill, 2000fb
Modeled Area Length, x (ft)	2,700	2,250	1,160	1,800	Estimated from concentration data and interpolation scheme (CH2M Hill, 2000b)
Source Type	Single Planar, Constant Source	Single Planar, Constant Source	Single Planar, Constant Source	Single Planar, Constant Source	Estimated, assume no vertical dispersion, fully penetrating source thickness (Aziz et al., 2000)
Source Thickness in Saturated Zone (ft)	12	12	12	12	Estimated from groundwater level data (CH2M Hill, 2000a, 2000f)
Source Width (ft)	1,500	1,500	1,000	1,500	Estimated, coincides with model dimensions (Aziz et al., 2000).
Source Concentration, C_1, C_2 (mg/L)	0.200	0.012	0.032	0.012	Analytical results (CH2M Hill, 2000f).

specific distribution coefficients (K_{oc}), the fraction organic carbon (f_{oc}), soil bulk density (ρ_b), and n_e.

K_{oc} values of 284 liters per kilogram (L/kg) and 118.5 L/kg were used for PCE and TCE, respectively. An f_{oc} value for Fluvial aquifer soils of 0.0029, a ρ_b of 1.67 kg/L, and an n_e of 0.30 results in computed R values for PCE and TCE of 5.5 and 2.9, respectively (see Table 5.1). Therefore, if V_x is 28.1 ft/yr, the transport velocities of PCE and TCE are on the order of 5.1 ft/yr and 9.7 ft/yr, respectively.

Additional simulations were performed for Flowpaths F1, F2, and F4 using a computed R value for PCE of 1.63. This value is based on an f_{oc} value for Fluvial aquifer soils of 0.0004 (Table 5.1). Therefore, a range of potential plume migration scenarios was simulated.

7.1.5.4 Biodegradation

BIOCHLOR models the reduction of chlorinated solutes according to a sequential first-order rate function (λ). The solute half-life represents the time in years for dissolved solutes to degrade by one-half as they migrate through the aquifer. Half-lives for PCE and TCE computed using the method of Buscheck and Alcantar (1995) were 28 years and 13 years, respectively (Table 5.1). These values are somewhat more conservative than the values of 7.7 years (PCE) and 3.9 years (TCE) estimated by CH2M Hill (2000f), and were computed using R values for PCE and TCE of 5.5 and 2.9, respectively.

Additional simulations were performed for Flowpaths F1, F2, and F4 using a PCE half-life of 7.7 years, which is representative of the decay rate computed using the method of Buscheck and Alcantar (1995) and a R value of 1.63 (Table 5.1). Therefore, a range of potential plume migration scenarios was simulated.

7.1.5.5 Other Model Input Data

The simulation time for the predictive models was arbitrarily set at 100 years. PCE/TCE model areas and source widths were estimated based on kriged March 2000 plume concentrations presented in CH2M Hill (2000f) (Figures 7.1 and 7.4). The kriged and contoured plume data resulted in dimensions for F1 being 1,500 feet wide by 2,700 feet long (MW47 to MW39). The F2 model domain was 1,000 feet wide by 2,250 feet long (MW39 to MW34), the F3 model domain was 1,000 feet wide by 1,160 feet long (MW62 to MW34), and the F4 model domain was 500 feet wide by 1,800 feet long. A source thickness of 12 feet was used for all four simulated flowpaths using water-level data from recent measuring events (CH2M Hill, 2000a and 2000f).

7.1.6 Model Simulations

Construction of calibrated models was not attempted due to the unknown location(s) and histories of the contaminant sources. Instead, the following simulations were run to estimate potential future contaminant migration toward wells MW34 and MW24:

1. Flowpath F1: The model was used to predict the maximum PCE concentration that could occur at MW39 within a 100-year time frame given a constant-

strength source of 200 µg/L in the vicinity of PZ04. This is the maximum concentration of PCE that has been detected near the southwestern corner of the MI.

2. Flowpath F2: Using the results from the F1 simulation described above, the model was used to predict the maximum PCE concentration that could occur at MW34 within a 100-year time frame given a constant-strength source of 12 µg/L at MW39.

3. Flowpath F3: The model was used to predict the maximum TCE concentration that could occur at MW34 within a 100-year time frame given a constant strength source of 32 µg/L at MW62. This is the TCE concentration detected at this well in March 2000.

4. Flowpath F4: The model was used to predict the maximum PCE concentration that could occur at MW24 within a 100-year time frame given a constant-strength source of 12 µg/L at MW39.

7.1.7 Model Results

The results of the four predictive simulations are described in this section. Model input and output summary sheets are included in Appendix C.

7.1.7.1 Simulation F1

Under a constant-strength source of 200 µg/L, the maximum predicted PCE concentration at MW39 after 100 years of transport was 0 µg/L assuming values for R and contaminant half-life of 5.5 and 28 years, respectively (i.e., the PCE plume emanating from near the southwestern corner of the MI was not predicted to reach MW39) (Table 7.2). Using values for R and half-life of 1.63 and 7.7 years, respectively, the maximum predicted PCE concentration at MW39 was 0.6 µg/L. Therefore, the PCE concentration detected at MW39 in March 2000 (12 µg/L) was used as the starting concentration for the Flowpath F2 simulation.

7.1.7.2 Simulation F2

Using a 12-µg/L constant-strength PCE source at MW39, the predicted PCE concentration at MW34 after 100 years was 0 µg/L (i.e., the PCE plume was not predicted to reach MW34) (Table 7.2). The maximum-predicted transport distance of PCE at a concentration of 5 µg/L along Flowpath F2 ranged from 300 to 530 feet. As described in Section 7.1.5.1, the contaminant velocity used in these simulations was the same as that used for Flowpath F1. If the steeper hydraulic gradient between MW39 and MW34 indicated on Figure 2.7 is used in the simulations (0.0155 ft/ft), then the maximum PCE concentrations predicted to migrate to MW34 within a 100-year timeframe ranged from 3 to 5 µg/L (Table 7.2).

TABLE 7.2
SUMMARY OF BIOCHLOR MODELING RESULTS
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHISE, TENNESSEE

Simulation Number	Flowpath	Modeled Contaminant	Flowpath Length (ft)	Retardation Coefficient	Decay Coefficient ($year^{-1}$)	Half-Life (years)	Hydraulic Gradient (ft/ft)	Plume Migration Distance (ft)[a/]	Concentration at End of Flowpath (µg/L)[b/]
F1-1	F1	PCE	2700	5.5	0.025	28	0.0037	1261	0
F1-2	F1	PCE	2700	1.63	0.090	7.7	0.0037	1576	0.6
F2-1	F2	PCE	2250	5.5	0.025	28	0.0037	530	0
F2-2	F2	PCE	2250	1.63	0.090	7.7	0.0037	313	0
F2-3	F2	PCE	2250	5.5	0.025	28	0.0155	2150	4.6
F2-4	F2	PCE	2250	1.63	0.090	7.7	0.0155	1254	2.6
F3-1	F3	TCE	1160	2.9	0.053	13	0.0037	1005	3.5
F3-2	F3	TCE	1160	2.9	0.053	13	0.0344	>1160	25
F4-1	F4	PCE	1800	5.5	0.025	28	0.0007	114	0
F4-2	F4	PCE	1800	1.63	0.090	7.7	0.0007	115	0

[a/] Plume migration distance represents the furthest downgradient extent of the 5 microgram per liter concentration.

[b/] µg/L - micrograms per liter.

7.1.7.3 Simulation F3

Because the PCE and TCE plume geometries differ, the fate and transport of TCE was modeled alone, ignoring mass transfer contributions from degradation of PCE. Based on an existing constant-source concentration at MW62 of 32 µg/L, the maximum predicted concentration of TCE at MW34 during the 100-year simulation period was 4 µg/L (Table 7.2). The contaminant velocity used for F1 and F2 also was used in the F3 simulation because measured water levels at MW62 and MW34 did not appear representative of the velocity field that created the observed TCE plume. However, if the steeper hydraulic gradient between MW62 and MW32 indicated on Figure 7.2 is used in the simulation (0.0344 ft/ft), then the maximum TCE concentration predicted to migrate to MW34 within a 100-year timeframe is 25 µg/L.

7.1.7.4 Simulation F4

Using a 12-µg/L constant-strength PCE source beginning at MW39, the predicted concentration at MW24 after 100 years was 0 µg/L (i.e., the PCE plume was not predicted to reach MW24) (Table 7.2). The maximum predicted transport distance of PCE at a concentration of 5 µg/L was slightly greater than 100 feet downgradient from MW39. The contaminant velocities used in these simulations were calculated using the same combinations of hydraulic conductivity, effective porosity, and retardation coefficient used for simulations F1 and F2. However, the hydraulic gradient was calculated using the water table elevation data for MW39 and MW24 shown on Figure 7.5.

7.1.8 Discussion of Results

The modeling results presented in Section 7.1.7 indicate that the PCE plume in the southwestern corner of the MI will not migrate to the immediate vicinity of MW34 or to the Depot boundary at well MW24 at concentrations of concern after 100 years of transport. Therefore, despite the limited nature of PCE biodegradation in MI groundwater (Section 5), the model indicates that this plume will not pose a threat to groundwater quality in deeper zones (e.g., the sands of the Jackson/Upper Claiborne Group and the Memphis Aquifer), to the Allen Well Field, or to groundwater quality south of the Depot.

The conclusions presented for PCE in the previous paragraph also pertain to TCE. It appears likely that MW34 is screened within the sands of the Jackson/Upper Claiborne Group, below the base of the Fluvial aquifer (Figure 2.5). Therefore, the hydraulic gradient along Flowpath F3, conservatively calculated using water levels measured in this well, does not appear to be representative of the gradient within the Fluvial aquifer along the entire length of this flowpath, and the resulting model prediction is conservative (i.e., the maximum TCE concentration that could migrate to MW34 likely is less than 25 µg/L). Given the results of the BIOSCREEN modeling performed by CH2M Hill (Section 7.1.3), it is unlikely that the TCE plume in the southwestern corner of the MI will pose a significant threat to the Allen Well Field.

The modeling results indicate that the primary remedial objective of the enhanced bioremediation program outlined in the draft ROD for the MI (CH2M Hill, 2001a) should be to reduce relatively elevated CVOC concentrations below cleanup goals within the desired timeframe, rather than to prevent or minimize future plume migration (although this will be the long-term effect of reducing maximum CVOC concentrations). Therefore, injection of nutrients/chemicals should be focused in plume hot spots.

7.2 STATISTICAL ANALYSIS OF MAIN INSTALLATION PLUME STABILITY

In addition to the BIOCHLOR modeling described in Section 7.1, statistical tests were performed to assess the stability of the chlorinated solvent plumes in the southwestern corner of the MI. The statistical algorithms used to accomplish this objective included the MAROS software package (AFCEE, 2000) and an alternate algorithm developed by Parsons ES.

7.2.1 Methods of Analysis

Plume dynamics can be evaluated by examining changes in the areal distribution of contaminants through time, or by examining changes in the concentrations of contaminants through time at individual well locations within or downgradient from a plume. Temporal data (chemical concentrations measured at different times) can be examined visually (graphically) or with statistical tests to evaluate plume stability. If removal of chemical mass is occurring in the subsurface as a consequence of attenuation processes or operation of an engineered remediation system, mass removal will be apparent as a decrease in chemical concentrations through time at a particular sampling location, as a decrease in chemical concentrations with increasing distance from chemical source areas, and/or as a change in the suite of chemicals through time or with increasing migration distance (e.g., an increase in biodegradation daughter products along the plume flowpath).

Temporal chemical concentration data can be evaluated by plotting contaminant concentrations through time for individual monitoring wells, or by plotting contaminant concentrations versus downgradient distance from the contaminant source for several wells along the groundwater flowpath over several monitoring events. Plotting temporal concentration data is recommended for any analysis of plume stability (Wiedemeier and Haas, 1999); however, visual identification of trends in plotted data may be a subjective process, particularly (as is often the case) if the concentration data do not have a uniform trend, but are variable through time (Figure 7.6).

The possibility of arriving at incorrect conclusions regarding plume stability on the basis of visual examination of temporal concentration data can be reduced by examining temporal trends in chemical concentrations using various statistical procedures, including regression analyses and the Mann-Kendall nonparametric test for trends. The Mann-Kendall test (Gibbons, 1994) is well suited for application to the evaluation of environmental data because the sample size can be small (as few as four data points), and no assumptions are made regarding the underlying statistical distribution of the data. The Mann-Kendall test statistic can be calculated at a specified level of confidence to evaluate whether a temporal trend is present in contaminant concentrations detected through time in samples from an individual well. If a trend is determined to be present, a non-parametric slope of the trend line (change per unit time) can also be estimated using the test procedure. A negative slope (indicating decreasing contaminant concentrations

Figure 7.6 Conceptual Representation of Temporal Trends and Temporal Variation in Concentrations

through time) or a positive slope (indicating increasing concentrations though time) provides statistical confirmation of temporal trends that may have been identified visually (Figure 7.6). The trend line slopes calculated using MAROS software are contained in the MAROS output (Appendix C).

Two methods were used to perform Mann-Kendall analyses on DDMT contaminant data. The first method used the MAROS software developed by Groundwater Services, Inc. for AFCEE. The second method used a Geographical Information System (GIS)-based algorithm developed by Parsons ES. Both analysis methods, the justification for use of two methods, and the analysis results are described in the remainder of this section. The data set used for the statistical analysis of the chlorinated solvent plumes at the MI included groundwater monitoring event results from March 1989 through March 2000. Both TCE and PCE data were included in the analysis.

7.2.2 MAROS Analysis

7.2.2.1 Description of MAROS Tool

The MAROS software consists of a set of small programs (macros) that operate within an electronic database environment (Microsoft™ Access97®) and perform certain mathematical or statistical functions using data that have been loaded into the database. MAROS makes extensive use of graphical user interfaces, and is generally a user-friendly tool. MAROS appears to have been developed primarily to assist non-technical personnel (e.g., facility environmental managers) in the organization, preliminary evaluation, and presentation of monitoring data. Application of the MAROS tool to the site-specific evaluation is dependent upon the amount and quality of the available data (e.g., data requirements for a temporal trend analysis include a minimum of four distinct sampling events).

MAROS can be used to assess plume stability and to evaluate and optimize a groundwater monitoring program. Evaluating plume stability is one aspect of optimizing a remedial process. Given the current lack of a well-defined groundwater monitoring program for the MI, the sole objective of the MAROS evaluation was to evaluate the stability of the PCE and TCE plumes in the southwestern portion of the MI. Therefore, the discussion of MAROS simulation results is limited to plume stability considerations. However, a complete MAROS simulation was performed, and the output is contained in Appendix C.

7.2.2.2 MAROS Temporal Trend Results for TCE

MW21 and MW39 are the monitoring points that exhibit increasing concentration trends for TCE (red color-coding in Figure 7.7). MW21 is located within or near a potential source area near the southwestern corner of the MI. An increasing trend at this location, if it continues, suggests the potential for plume expansion in the future as the higher concentrations migrate downgradient. MW39 is located at the inferred perimeter of the plume; the observed increasing trend at this location also is indicative of a potentially expanding plume.

Figure 7.7 Summary of MAROS Trend-Analysis Results for Main Installation TCE Plumes

MAROS indicates a decreasing trend at well MW41; however, no detected concentrations of TCE have been reported for this location. The MAROS tool assigns a value equal to a percentage of the laboratory detection or reporting limit to analytical results reported as "not detected." This convention potentially can generate misleading results in the temporal evaluation of monitoring data from a particular monitoring point. For example, analytical methods and protocols have undergone a number of changes through the years, and these improvements have generally resulted in lower detection limits over time. Consistent substitution of a positive value when an analyte was not detected (as the MAROS software does) can result in the identification of an apparent decreasing temporal trend in chemical concentrations through time, when in fact no such trend exists. In such cases, the "trend" is an artifact of decreasing analytical detection limits through time.

According to MAROS, TCE analytical data for several wells outside of the 5-µg/L TCE plume concentration contour (MW23, MW34, and MW38) have no statistical trend (Figure 7.7). These wells have either not contained detectable concentrations of TCE, or have sporadically contained only trace levels of this compound. The lack of TCE detections at MW23 suggests a lack of significant plume expansion in this cross-gradient direction (south-southeast). Wells MW34 and MW38 appear to be screened in the sand aquifer underlying the Fluvial aquifer, and the MAROS results indicate that this aquifer has not been significantly impacted by the TCE plume.

According to MAROS, wells MW19, MW20, MW47, MW48, and MW55 exhibit stable trends (Figure 7.7). However, MW20 is the only well at which TCE has been previously detected. Therefore, the stable trends exhibited in wells MW19, MW47, MW48, and MW55 are due to the fact that TCE has not been detected at these locations. Because fewer than four analytical results are available for several other wells, no statistical determination can be made regarding the presence or absence of temporal trends in TCE concentrations at these locations.

7.2.2.3 MAROS Temporal Trend Results for PCE

MW21 and MW39 are the monitoring points that display increasing concentration trends for PCE (Figure 7.8). Available data suggest that these two wells are located along the approximate axis of the primary PCE plume, and therefore plume expansion parallel to the dominant groundwater migration direction cannot be ruled out at this time. Similar to TCE, the decreasing trend indicated for PCE at MW41 appears to be an artifact of decreasing detection/reporting limits over time, because PCE has never been detected at this location.

According to MAROS, the temporal PCE concentration data for wells MW20, MW23, MW34, MW38, and MW47 do not have a statistical trend (Figure 7.8). With the exception of MW47, these wells are outside of the 5-µg/L concentration contours and have either contained only sporadic trace PCE concentrations, or have never had detectable PCE concentrations. PCE concentrations at MW47 have ranged from 1 to 200 µg/L, with no consistent temporal trend.

MAROS identified stable PCE concentration trends at wells MW19, MW48, and MW55. However, none of these wells have ever had detectable concentrations of PCE.

Figure 7.8 Summary of MAROS Trend-Analysis Results for Main Installation PCE Plumes

Therefore, the trends in these wells are merely artifacts of stable detection/reporting limits over time. Fewer than four analytical results are available for several other wells, and no statistical determination can be made regarding the presence or absence of temporal trends in PCE concentrations at these locations.

7.2.3 Alternate Algorithm Analysis

7.2.3.1 Description of the Alternate Algorithm

As described above, examination of the structure and function of the MAROS tool identified potential limitations on its usefulness in groundwater monitoring evaluations. For example, inspection of the summary statistics for the groundwater monitoring data at the MI indicates that a significant percentage of the results are based on values that were reported as “not detected.” MAROS assigned a positive value to these results, which caused erroneous trends (e.g., decreasing trends) to be identified. The alternate algorithm corrects this problem by maintaining the “not detected” designation. The significance of temporal trends and the results of this alternate analysis are described in the following subsections.

7.2.3.2 Alternate Algorithm Results

Similar to the MAROS tool, the alternate algorithm identifies temporal trends using the Mann-Kendall test. The objective of the temporal trend evaluation for PCE and TCE was to assess plume stability.

Summary results of Mann-Kendall temporal trend analyses for PCE and TCE are presented in Table 7.2. As implemented, the algorithm used to evaluate trends assigned a trend of “Not Detected” in cases where analytes were consistently not detected through time, rather than using detection-limit values that could generate potentially misleading and anomalous “trends” in concentration. Color-coding of the table entries denotes the presence/absence of temporal trends, and allows those monitoring points having no detectable concentrations, decreasing or increasing concentrations, or no discernible trend in concentrations to be readily identified.

7.2.3.3 Alternate Algorithm Temporal Trend Results for TCE

Monitoring points displaying increasing concentration trends for TCE (red color-coding in Table 7.3) include MW21 and MW34 (Figure 7.9). The MAROS tool also identified an increasing trend at MW21, which is located in or near a potential source area. The increasing trend identified for MW34 is suspect (MAROS classified this well as “no trend”). This well is screened in the sand aquifer underlying the Fluvial aquifer, and this lower zone has been relatively unaffected by groundwater contamination in the Fluvial aquifer. TCE was detected in one of seven samples collected from this well from 1989 through 1998; the single detection was 1 μg/L in 1996. In contrast, TCE has been detected in four of the six most recent samples collected in 1999 and 2000 at concentrations ranging from 0.85 μg/L to 4.39 μg/L. The greater frequency of TCE detections during the past 2 years may be a result of improved sampling methods and/or analytical procedures rather than an actual increase in the concentration of this analyte; however, additional samples from this well should be collected to confirm this

TABLE 7.3
SUMMARY OF ALTERNATE ALGORITHM TEMPORAL-TREND ANALYSIS FOR THE MAIN INSTALLATION

REMEDIAL PROCESS OPTIMIZATION

DEFENSE DEPOT MEMPHIS, TENNESSEE

WELL	TRICHLOROETHENE	TETRACHLOROETHENE
HY01	< 4 meas[a/]	< 4 meas
HY02	< 4 meas	< 4 meas
HY07	< 4 meas	< 4 meas
HY09	< 4 meas	< 4 meas
MW19	ND	ND
MW20	no trend	ND
MW21	+[d/]	+
MW22	no trend	ND
MW23	ND	no trend
MW24	ND	ND
MW34	+	no trend
MW38	no trend	ND
MW39	no trend	+
MW41	ND	ND
MW47	ND	ND
MW48	ND	ND
MW55	ND	ND
MW62	< 4 meas	< 4 meas
MW66	< 4 meas	< 4 meas
MW72	< 4 meas	< 4 meas
PZ01	< 4 meas	< 4 meas
PZ03	< 4 meas	< 4 meas
PZ04	< 4 meas	< 4 meas
PZ08	< 4 meas	< 4 meas

[a/] < 4 meas	= Less than four measurements at the monitoring well; trend analysis not performed.
[b/] no trend	= No statistically significant temporal trend in concentrations.
[c/] ND	= Constituent has never been detected at well monitoring.
[d/] +	= Statistically significant increasing trend in concentration.

Figure 7.9 Summary of Alternative Algorithm Trend-Analysis Results for Main Installation TCE Plume

observation. In contrast, TCE has not been detected at nearby well MW38, which is also screened beneath the base of the Fluvial aquifer.

Several wells located near the inferred cross-gradient perimeter of the TCE plume (MW20, MW22, MW39, and HY01) also have no statistical trend (Figure 7.9), indicating that the plume is neither expanding nor receding at these locations. Fewer than four analytical results are available for several other wells, and no statistical determination can be made regarding the presence or absence of temporal trends in TCE concentrations at these locations.

7.2.3.5 Alternate Algorithm Temporal Trend Results for PCE

Similar to the MAROS results described in Section 7.2.2.3, monitoring points displaying increasing concentration trends for PCE include MW21 and MW39 (Figure 7.10). These trends indicate a potentially expanding plume. The results of the alternate algorithm indicate that PCE data for wells MW23, MW34, and HY01 do not have a statistical trend. The lack of trends for MW23 and HY01 indicate a lack of plume expansion to the south; MW34 is located approximately 1,500 feet from the inferred location of the 5-µg/L PCE concentration contour, and the lack of a trend at this well is not significant in terms of the plume stability analysis.   Fewer than four analytical results are available for several other wells, and no statistical determination can be made regarding the presence or absence of temporal trends in PCE concentrations at these locations.

7.3 SUMMARY OF PLUME STABILITY EVALUATION

Based on the results of the BIOCHLOR modeling (Section 7.1) and the statistical analyses (Section 7.2), further downgradient expansion of the PCE and TCE plumes in the southwestern corner of the MI cannot be ruled out based on increasing concentration trends at selected wells in the plume. However, plume expansion, if it occurs, should not pose a significant risk to off-site receptors (i.e., the Allen Well Field or potential receptors south of well MW24).

Figure 7.10 Summary of Alternate Algorithm Trend-Analysis results for Main Installation PCE Plumes

SECTION 8

REVIEW OF GROUNDWATER MONITORING PROGRAM

The current groundwater monitoring program for Dunn Field was evaluated to identify potential opportunities to streamline monitoring activities while maintaining an effective program that monitors the performance of the groundwater extraction system and the potential for contaminants to migrate beyond the system. The approach used in this evaluation was discussed in Section 4.1, and is summarized on Figure 4.1. This approach involves evaluating the importance of each well in the monitoring network and its sampling frequency by using a combination of qualitative and statistical (temporal and spatial) analyses.

The Dunn Field monitoring program consists of 20 monitoring wells that are sampled quarterly for analysis of VOCs, and annually for analysis of SVOCs, metals, and organochlorine pesticides (see Figure 8.1 for well locations). Regularly-sampled groundwater extraction wells were not included in this evaluation. For this evaluation only monitoring results for TCE were considered, because TCE is the predominant contaminant present in groundwater at Dunn Field, and it has the greatest influence on the monitoring requirements. Therefore, the conclusions derived from the evaluation are preliminary and are presented as an example of the recommended evaluation approach. Other COCs should be included in this evaluation in the future for a more complete evaluation of the monitoring program.

To prepare for this evaluation, relevant site data were reviewed and spatial and temporal statistical analyses were performed. The site data reviewed includes the following:

- Description of the hydrostratigraphy,
- Location of the source(s),
- Location of potential receptors and POCs,
- Configuration of the contaminant plume (vertically and laterally),
- TCE concentration database,
- Directions and rates of contaminant movement,
- Design of remedial system, and
- Monitoring well locations and completion zones.

Figure 8.1 Results of Temporal-Trend Analyses for TCE in Dunn Field Groundwater

A temporal analysis of TCE concentrations dating from 1989 to March 2, 2000 was conducted using the Mann-Kendall non-parametric test for trends. The method for applying the Mann-Kendall test was the "alternate algorithm" used to assess plume stability at the MI (Section 7.2). The temporal analysis identified the TCE concentration trends for each well (i.e., whether TCE concentrations exhibit an increasing trend, a decreasing trend, no significant trend, or were consistently below the laboratory detection limit). This information helps in evaluating the importance of one well over another in the monitoring network. The results of the temporal trend analysis for each well are shown on Figure 8.1.

A spatial statistical kriging analysis of the type described in Section 4.1.2 was performed to assist in evaluating the well locations in the monitoring network. Although this approach can be used to assess both redundancy and gaps in monitoring point locations, only redundancy was considered for this evaluation because installation of additional wells is planned in the future. The importance of each well was evaluated by successively removing the well from simulations and evaluating if a significant loss of information (represented by increases in standard deviations) occurred as a result of excluding the well from the network. If exclusion of a well resulted in a change in the global kriging standard deviation of less than approximately 1 percent, then the well was not considered spatially important, from a statistical standpoint.

The importance of each well and its sampling frequency were evaluated using knowledge of site conditions, results of the statistical analyses, and professional judgment. To facilitate the evaluation process, the 20 monitoring wells were divided into 4 groups based on where the well is located with respect to the plume. These four groups were selected because the wells within these groups share similar monitoring objectives. The four groups include wells that are located:

- Hydraulically upgradient from the TCE plume (to monitor background water quality),

- Within the TCE plume (to monitor plume boundaries and performance of the remedial system),

- Hydraulically cross-gradient from and generally outside of the TCE plume (to monitor lateral plume boundaries over time), and

- Hydraulically downgradient of the TCE plume (to monitor the potential for plume expansion in a downgradient direction)

Wells were identified in each of the above groups that would best achieve the general monitoring objective for the well group. The wells identified as least important were recommended to be excluded from the monitoring network or sampled less frequently. The results of the monitoring program evaluation for Dunn Field, including the results of the temporal and spatial analyses, are summarized in Table 8.1. These results are discussed in more detail below.

TABLE 8.1
RESULTS OF MONITORING PROGRAM EVALUATION FOR DUNN FIELD
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Well No.	Temporal Trend[a/]	Spatially Strategic?[b/]	Include or Exclude?	Recommended Sampling Frequency	Rationale for Continuing or Discontinuing Sampling
Wells Located Hydraulically Upgradient From TCE Plume:					
MW-51	decreasing	yes	Include	annual	Low TCE concentrations (up to 15 µg/L) detected during 11 samping rounds over past 5 years; indicates background water quality contains low levels of TCE.
Wells Located Within TCE Plume:					
MW-15	increasing	yes	Include	semi-annual	Monitors performance of extraction system; increasing trend does not pose a risk because well is presumably within extraction system capture zone.
MW-31	decreasing	no	Include	semi-annual	Defines lateral extent of TCE plume to the north; monitors performance of extraction system.
MW-32	no trend	yes	Include	semi-annual	Monitors performance of extraction system and magnitude of off-site contamination.
MW-54	decreasing	no	Include	semi-annual	Monitors performance of the extraction system and downgradient boundary of plume.
MW-57	ID	yes	Include	semi-annual	Defines lateral extent TCE plume to the south; monitors performance of extraction system.
MW-59	ID	yes	Exclude		Redundant with MW-68 (they are located within 200 feet of each other and screened in similar intervals); low TCE concentrations (up to 4 µg/L) exist at MW-59 and well is presumably located within capture zone of extraction system.
MW-68	ID	no	Include	semi-annual	Monitors performance of extraction system.
MW-69	ID	yes	Include	semi-annual	Monitors performance of extraction system.
MW-70	ID	yes	Include	semi-annual	Monitors performance of extraction system; located in most contaminated portion of plume.
MW-71	ID	yes	Include	semi-annual	Monitors performance of extraction system.
Wells Located Hydraulically Cross-Gradient of and Generally Outside of the TCE Plume:					
MW-14	no trend	yes	Include	semi-annual	Defines the lateral extent of the TCE plume to the south
MW-34	increasing	yes	Include	semi-annual	Monitors potential vertical migration of TCE.
MW-40	BDL	yes	Exclude		TCE not detected in 12 sampling rounds over past 4 years; well is 800 to 900 feet cross-gradient from plume boundary.
MW-30	BDL	yes	Exclude		TCE not detected in 14 sampling rounds over past 11 years.
MW-56	ID	yes	Include	semi-annual	Monitors lateral extent of the plume.
MW-58	ID	yes	Exclude		Redundant with well MW-56 (located within 200 feet of each other and screened in similar intervals).

TABLE 8.1 (Continued)
RESULTS OF MONITORING PROGRAM EVALUATION FOR DUNN FIELD
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Well No.	Temporal Trend[a/]	Spatially Strategic?[b/]	Include or Exclude?	Recommended Sampling Frequency	Rationale for Continuing or Discontinuing Sampling
Wells Located Hydraulically Downgradient of the Outermost Extent of the TCE Plume:					
MW-33	BDL	yes	Include	annual	TCE not detected in 12 sampling rounds over past 8 years; continue monitoring less frequently to assess potential changes in concentration resulting from extraction operations and potential contaminant bypass of the system.
MW-44	no trend	yes	Include	semi-annual	Defines downgradient extent of TCE plume; monitors potential bypass of extraction system.
MW-67	ID	yes	Include	annual	Monitors potential vertical migration of contamination.

Note: For newly installed monitoring wells, the sampling frequency shown above is recommended after quarterly sampling has been conducted for one year to establish baseline condition.

[a/] " ID" indicates there are insufficient data to assess statistical trends in TCE concentrations.
"BDL" indicates TCE concentrations were below the laboratory detection limit for all samples analyzed.
"increasing" indicates there is a statistically significant increasing trend in TCE concentrations.
"decreasing" indicates there is a statistically significant decreasing trend in TCE concentrations .
"no trend" indicates there is not a statistically significant trend in TCE concentrations.

[b/] Well is considered spatially strategic if exclusion of well from monitoring program results in a change in the global kriging standard deviation greater than approximately 1 percent.

8.1 RESULTS OF MONITORING NETWORK EVALUATION

Based on the results of this evaluation, the majority of the 20 wells being monitored at Dunn Field are appropriate for inclusion in the monitoring network. This is because the current groundwater extraction system is interim, and the data collected from the wells will be used to select and design a final remedy. Only 4 of the 20 wells should be considered for exclusion from the monitoring program based on historical TCE concentrations. These four wells are MW30, MW40, MW58, and MW59 (Figure 8.1).

Well MW30 is located hydraulically cross-gradient of the TCE plume, and TCE has not been detected during any of the 14 samples collected from this well over the past 11 years. It is reasonable to conclude that TCE at that location will remain below detection limits in the future; hence the well is recommended for exclusion from the network.

TCE has not been detected in well MW40 in 12 sampling rounds over the past four years; this well is located approximately 800 to 900 feet cross-gradient from the boundary of the plume. Well MW40 is recommended for exclusion from the monitoring network because it is not particularly useful for defining the lateral boundary of the plume or for detecting potential future migration of contaminants. However, this decision should be reevaluated based on recently collected hydrogeologic and chemical data.

Well MW58 is recommended for exclusion from the monitoring network because it appears to be providing data redundant with well MW56, which is located only 200 feet away and is screened at a similar depth. Wells MW56 and MW14 are adequate for monitoring the lateral extent of the plume at that location.

Well MW59 is recommended for exclusion from the monitoring network because it is providing redundant information with well MW68. These wells are located within 200 feet of each other and are screened at similar depths. Well MW68 is more important for evaluating the performance of the extraction system because TCE concentrations at well MW68 (45 μg/L) are higher than TCE concentrations at well MW59 (4 μg/L or less).

Although three wells were not considered to be spatially strategic from a statistical standpoint (MW31, MW54, and MW68), they are recommended for inclusion in the monitoring network based on the qualitative evaluation. These wells are important for defining the boundary of the plume and/or for monitoring the performance of the extraction system.

8.2 RESULTS OF THE SAMPLING FREQUENCY EVALUATION

Quarterly sampling of each newly-installed monitoring well for one year is recommended to establish baseline conditions. After baseline water quality conditions have been established, sampling frequencies should be selected to adequately identify potential changes in concentrations that may require an action. For example, if contaminants are found to bypass the groundwater extraction system, then the extraction system may require modification to correct the situation. Therefore, wells used to monitor potential bypass of contaminants should be sampled at a frequency that allows for timely correction of the system. In addition wells located closer to the extraction system should be sampled more frequently than those located farther from the system.

This is because evidence (or lack of evidence) of remediation will likely be observed first in wells closer to the system. Once groundwater at a well has been influenced by remediation, it may be appropriate to decrease the sampling frequency at that well and increase the sampling frequency for wells located farther downgradient.

The groundwater seepage velocity at Dunn Field is moderate, ranging from 0.17 ft/day to 0.25 ft/day (62 to 91 ft/yr) (CH2M HILL, 2000f). Therefore, dramatic changes in contaminant concentrations at a particular location due to plume movement are not expected to occur very rapidly. Thus, it is recommended that the sampling frequency be reduced to semi-annual or annual. Annual sampling is recommended for wells installed in locations where groundwater quality is not expected to change significantly between monitoring events. These wells include background well MW51, well MW33 (because TCE concentrations have consistently been below laboratory detection limits over the past 8 years) and well MW67 (because it is screened nearly 200 feet below the alluvial aquifer).

The LTM program will likely evolve over time as additional site characterization results are obtained and a final remedy is selected and implemented. The qualitative, temporal, and spatial evaluations described in this section and in Section 4.1 should be repeated following completion of site characterization activities; these evaluations should incorporate data for all groundwater COCs. Less-frequent monitoring of wells located outside of the plume may be appropriate in the future as the capture zone of the groundwater extraction system becomes better defined. At that time, monitoring efforts should be focused to a greater degree on evaluating the performance of the remedial system.

8.3 TARGET ANALYTE LIST

Groundwater samples collected at Dunn Field as part of the O&M for the Groundwater Interim Remedial Action extraction system have been analyzed quarterly for VOCs, and annually for metals, SVOCs, and organochlorine pesticides. Parsons ES did not obtain all of the groundwater quality data collected to date under this monitoring program. However, based on the data that were obtained, the following preliminary observations and recommendations regarding the target analyte list are offered:

- Organochlorine pesticides have generally not been detected in groundwater samples, and deletion of these analytes from further consideration should be considered unless sampling of new wells in less-characterized areas is initiated.

- Detections of SVOCs have been limited to only several of the numerous compounds targeted by Method SW8270. The primary semivolatile analyte of concern appears to be bis(2-ethylhexyl)phthalate in that detected concentrations have exceeded the USEPA Region III risk-based concentration for tap water. This compound is a plasticizer that is a constituent of most plastic apparatus and is also found in latex gloves; therefore, it may be introduced into the sample during sample collection, handling, or analysis. This is supported by the detection of this compound in equipment blanks as reported in OHM (2000). In addition, this compound may be ubiquitous in the environment rather than site-related; its presence in background groundwater quality samples should be assessed to determine if this is the case. Based on these observations, it may be possible to reduce or eliminate future sampling for SVOCs in well-characterized areas.

These observations should be re-evaluated in light of the full 2-year data set collected as part of the extraction system O&M, and appropriate modification of the target analyte list should be considered.

SECTION 9

DIFFUSION SAMPLING EVALUATION

Recent innovations in groundwater monitoring technologies have resulted in the development of PDB samplers as a cost-effective approach to monitoring VOCs at well-characterized sites, such as Dunn Field and the MI at DDMT. PDB samplers for VOC monitoring utilize passive sampling techniques that eliminate the need for well purging. These samplers are typically water-filled containers that are initially deployed within a screened interval of a well. Over an equilibration period (typically at least two weeks), the concentration of VOCs within the PDB sampler reaches equilibrium with VOC concentrations in the surrounding groundwater due to diffusion across a semi-permeable membrane. Following the equilibration period, the PDB sampler is retrieved from the well, and the water from within the diffusion sampler is transferred to a conventional sample container and submitted to a laboratory for analysis.

9.1 PDB SAMPLER DESCRIPTION

The United States Geological Survey (USGS) has developed a PDB sampler that is commercially available and that has undergone method development and field verification studies to assess comparability with more conventional groundwater sampling techniques (Vroblesky and Campbell, 2000). Results from this and other studies (e.g., McClellan AFB, 2000) have shown a close correlation in measured VOC concentrations between PDB samplers and conventional purge-and-sample approaches. PDB samplers have been successfully used at well-characterized sites. Multiple PDB samplers in a single well have been used to delineate the vertical distribution of groundwater contaminants, which provides information that is generally unavailable using conventional purging techniques (Hare *et al.*, 2000; McClellan AFB, 2000; Vroblesky and Campbell, 2000). However, there have been some concerns raised concerning the fundamental premise of PDB samplers, namely that there is sufficient flow through the well screen to allow aquifer equilibration with the water in the bag to occur. If flow is insufficient, potentially unrepresentative measurements may result (Barcelona, 2000; Hare *et al*., 2000). Typically, a side-by-side comparison using PDB samplers, followed immediately by conventional purge sampling and/or comparison with historical data, is performed during the initial deployment and evaluation of PDB samplers for use at a site.

The standard USGS diffusion sampler consists of a water-filled, low-density polyethylene bag, which acts as a semi-permeable membrane. The USGS sampler typically is constructed of a 1.5-foot-long section of a 2-inch-diameter, 40-mil polyethylene bag that is heat-sealed on both ends (Figure 9.1). The sampler holds approximately 300 milliliters (ml) of deionized, distilled water. A longer 3-inch-diameter

Figure 9.1 Example of Diffusion Sampler Deployment for Vertical Profiling

sampler that holds approximately 500 ml of water also is available if larger sample volumes are required. The sampler is placed in polyethylene mesh tubing for abrasion protection, attached to a weighted rope, and lowered to a predetermined depth within the screened interval of a well. The rope is weighted to ensure that the sampling devices are positioned at the correct depth and that they do not float upward through the water column. Following equilibration and upon recovery of the diffusion samplers from the wells, the samplers are cut open, and water samples are transferred into 40-ml volatile organics analysis (VOA) vials and submitted to a laboratory for analysis.

The VOC concentrations within the PDB sampler are impacted by the flow of groundwater through the depth interval where the sampler is deployed; thus, they result in "integrated" concentrations over time. Analyte concentrations are representative of the last portion of the deployment period (thought to be no more than approximately 2 weeks for most wells). Typically, one PDB sampler is deployed for each well that requires VOC monitoring. However, these samplers can also be placed end-to-end within a monitoring well to develop a vertical contamination profile. Since the PDB samplers provide a depth-specific measurement of VOCs, vertical profiling is generally recommended during the first PDB sampling event to assess the most appropriate depth interval for LTM.

Because of the equilibration time required for passive diffusion sampling, this method may not be suitable in cases where rapid assessment of groundwater quality is desired. Also, due to the relatively recent development of this methodology, site-specific comparisons with more conventional purging methods for groundwater sampling may be required. Therefore, the use of PDB samplers is best suited for LTM at sites that are already well-characterized.

9.2 USE OF DIFFUSION SAMPLERS AT DDMT

DDMT currently is performing groundwater monitoring at Dunn Field to monitor the performance of the groundwater extraction system in reducing groundwater contaminant concentrations. Quarterly monitoring for VOCs from 20 monitoring wells and 11 groundwater recovery wells is currently required, as well as annual sampling for VOCs, SVOCs, metals, and (for recovery wells only) pesticides/herbicides. It is anticipated that LTM for VOCs will also be required for the MI solvent plumes. Groundwater contamination at both Dunn Field and the MI is well characterized, with VOCs identified as the primary COCs. Therefore, DDMT is a good candidate for evaluation of PDB samplers to assess potential savings in effort and costs.

There are several potential advantages to using PDB samplers at DDMT:

- PDB samplers are easy to deploy and recover, resulting in quick and simple installation and sample collection. A minimal amount of field equipment is required.
- PDB samplers are relatively inexpensive.
- Minimal decontamination is required; only the scissors or knife used to open the PDB sampler bag requires decontamination between uses.

- Negligible quantities of investigation-derived waste (IDW) are produced; only the remaining sampler water and a minimal amount of decontamination fluids require disposal.

- Diffusion sampling requires little or no preparatory field work, except for measuring the rope length for the correct sampling depth prior to installation of the sampler in the well.

The following potential disadvantages were identified:

- The system is suitable only for collection of samples for VOC analysis. At Dunn Field, current monitoring requirements include quarterly sampling for VOC analysis, and annual sampling for additional parameters. Therefore, use of PDB samplers will not be sufficient for the annual sampling event unless requirements for analysis of these additional parameters are relaxed or deleted, and conventional purging methods will need to be used.

- The selectivity of the membrane also renders this method inappropriate for the measurement of commonly-used inorganic natural attenuation parameters (e.g. nitrate, ferrous iron, and sulfate). Additionally, measurement of typical field parameters (e.g. pH, conductivity, and ORP) is problematic using the diffusion sampler.

The deployment of PDB samplers within the groundwater recovery wells at Dunn Field may also be problematic due to the presence of pumps, piping and electrical lines within the recovery wells (this limitation also applies to conventional purging with submersible pumps or bailers). However, current practice for sampling these wells is to retrieve the sample from existing above-ground sampling points. This current practice is also quick and easily implemented, and thus use of PDB samplers at the recovery wells would not be advantageous. However, use of PDB samplers for monitoring wells at Dunn Field has potentially significant advantages; therefore, additional evaluation of the cost of this alternative approach was performed.

9.3 COST COMPARISON WITH CONVENTIONAL SAMPLING TECHNIQUES

Significant cost savings can be realized through the use of PDB samplers for VOC monitoring due to decreased labor and equipment costs in comparison to conventional purge sampling techniques (McClellan AFB, 2000). Estimated costs for PDB sampling were compared to conventional purge sampling costs at DDMT to quantify the potential cost savings associated with diffusion sampling (Table 9.1). This comparative estimate is based on sampling 20 monitoring wells for VOC analysis, as is currently required at Dunn Field. The following assumptions were incorporated into this cost estimate:

- Conventional purge sampling methods require the use of a gas-powered generator, a submersible pump assembly, and larger IDW containers; therefore a larger field vehicle is required than for PDB sampling. An additional $30 per day was applied to the purge sampling method to reflect the vehicle rental rate difference.

TABLE 9.1
COST COMPARISON OF CONVENTIONAL PURGE SAMPLING AND PASSIVE DIFFUSION BAG SAMPLERS
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

CONVENTIONAL PURGE

Unit	Reusable?	Cost per Unit	Unit	Number Required	Cost per Event
Field Vehicle Addition[a/ b/]	no	$ 30.00	Day	3	$ 90.00
Pump Rental [b/]	no	$ 350.00	Week	1	$ 350.00
Generator Rental [b/]	no	$ 75.00	Day	3	$ 225.00
Tubing[b/]	yes	$ 0.52	Feet	100	$ 52.00
Meter Rental (day rate)[b/c/]	no	$ 190.00	Week	1	$ 190.00
Drums for IDW	potentially	$ 50.00	Each	4	$ 200.00
Labor (1.5 hr/person/sample)[b/]	no	$ 75.00	Hour	60	$ 4,500.00
				Total Cost	$ 5,607.00
				Number of Samples Collected (each event)	20
				Cost per Sample	**$ 280.35**

PASSIVE DIFFUSION BAG SAMPLER

Unit	Reusable?	Cost per Unit	Unit	Number Required	Cost per Event
Diffusion Sampler [e/]	no	$ 14.00	Each	20	$ 280.00
Stainless Weight Hanger	yes	$ 7.50	Each	20	$ 150.00
Stainless Steel Weight	yes	$ 13.50	Each	20	$ 270.00
Polypropylene rope	yes	$ 0.07	Feet	2,000	$ 140.00
5-gallon bucket for IDW	potentially	$ 10.00	Each	1	$ 10.00
Labor (40 min/person/sample)	no	$ 75.00	Hour	27	$ 2,000.00
				Total Cost	$ 2,850.00
				Number of Samples Collected (each event)	20
				Cost per Sample	**$ 142.50**

[a/] Additional cost for larger vehicle.

[b/] Assumes purge and sample productivity of 6 wells per day, 9 hour day, 2 person crew.

[c/] Meters include pH, conductivity, dissolved oxygen, and oxidation/reduction potential.

[d/] Assumes 1 drum per 4 wells sampled.

[e/] PDB sampler with mesh covering.

COST SAVINGS OF USING PDB SAMPLER (PER EVENT): $2,757

COST SAVINGS OF USING PDB SAMPLER (PER SAMPLE): $138

- Costs for reusable PDB equipment such as sampler hangers and weights and rope were included in the cost, thereby overestimating the costs for all but the initial sampling event. However, these costs are generally minor in comparison to other labor and equipment costs. Equipment costs were obtained from Eon Products, Inc., a commercial supplier of PDB samplers.

- The estimate assumes one PDB sampler would be deployed per monitoring well, which is appropriate for long-term monitoring. However, the initial PDB sampling event would require additional expenses associated with initial deployment of multiple samplers (at discrete depths) within each well for vertical profiling, as well as field analysis for these additional samples as described in Section 9.4.

- Deployment of new samplers could be performed immediately after retrieval of the previous quarterly samples for analysis. In this way, the PDB samplers will remain in the well for a full three months before retrieval.

- Common sampling supplies (e.g., protective gloves, plastic sheeting, sample containers, etc.) were assumed to be the same for each method and therefore were not considered in the cost analysis.

- Laboratory analytical expenses and costs for analysis of QA/QC samples (i.e., field duplicates and trip blanks) were assumed to be the same for each method and therefore were not considered in the cost analysis.

- Only containment costs for IDW were included in the estimate; disposal costs were not quantified.

As presented in Table 9.1, the cost per sample using the PDB sampler was approximately $118, and the cost per sample using the conventional purge sampling method was approximately $280. Additionally, an estimated total of approximately 144 gallons of IDW would be generated per quarter using the conventional purge sampling method; (assuming purging of at least three casing volumes as opposed to micropurging); negligible volumes (less than 5 gallons) of IDW would be generated using the diffusive sampler. Other studies have also noted significant cost savings of PDB samplers over micropurge techniques (Guest *et al*., 2000).

It is estimated that the use of PDB samplers would result in cost savings of $3,260 per quarter (Table 9.1); however, this savings would only be applied for three quarters of the year due to additional non-VOC parameters required for annual monitoring. This translates to a potential cost savings of over $9,800 per year, assuming quarterly monitoring. The cost savings would be less significant for wells sampled less frequently.

9.4 RECOMMENDATIONS FOR USE OF DIFFUSION SAMPLERS

Field testing of PDB samplers is recommended for LTM of VOCs at DDMT based on the demonstrated effectiveness at other sites, and on the significant potential cost savings over conventional purge sampling methods. The initial PDB sampling event should include vertical profiling as described in Section 9.1, and a thorough site-specific evaluation of the comparability of PDB sampling results with results from the current purge sampling methods. This evaluation would consist of performing a side-by-side comparison of PDB sampling results with conventional sampling results collected during the same monitoring event. This initial verification would result in additional monitoring expenses for this first event (estimated to be approximately $50,000 for a work plan, vertical profiling of 20 wells [assumed average of 3 samples per well], and a results report comparing PDB sampling results with conventional results).

Specific recommendations for the initial PDB evaluation at Dunn Field include the following:

- Mobilize to DDMT at least 3 weeks prior to the annual sampling event for initial deployment of PDB samplers.

- For vertical profile characterization, deploy one 18-inch-long PDB sampler for each 3 feet of saturated well screen within each monitoring well included in the Dunn Field LTM program.

- During the annual sampling event (at least two weeks after initial deployment), retrieve the PDB samples from each well, and collect four 40-ml VOA vial samples from each PDB sampler. Perform field screening or on-site colorimetric analysis for selected chlorinated solvents on one vial from each sampler for vertical profile characterization. Field test kits can be obtained from Strategic Diagnostics, Inc. (QuickTest® kit for volatile organic halides) or from ORS Environmental Systems (AccuSensor® System for TCE).

- Based on the field screening results, select one representative (biased high) PDB sample interval from each well for offsite laboratory analysis. This selected sample interval would then be used for subsequent PDB sampling at the subject well.

- Following retrieval of PDB samples from a well, purge and sample the well using existing procedures. In addition to the VOC and non-VOC parameters required for annual sampling, collect an additional 40-ml VOA vial sample for field screening or on-site colorimetric analysis for chlorinated solvents (for comparison with PDB field screening results).

- Document field labor and equipment costs associated with both PDB sampling and conventional purge sampling to verify potential cost savings.

- Evaluate comparability of PDB samples and conventional purge sampling data, and assess site-specific advantages/disadvantages and costs for both methods. Document findings and provide recommendations for future monitoring events.

SECTION 10

EVALUATION OF REMEDIATION GOALS

A clear understanding of the goals and objectives of a remediation project is an essential step in the RPO process. An understanding of remediation goals established for a site is required to evaluate the merits of those goals in light of an evolving CSM and changes in regulatory approaches. A draft ROD establishing cleanup goals for the MI at DDMT was released in January 2001 (CH2M Hill, 2001a), and a Final ROD was signed in February 2001 (CH2M Hill, 2001b). Proposed clean-up levels have not been developed, and a ROD has not been signed for Dunn Field. The RPO process for DDMT provides an opportunity to promote interaction and communication among regulatory officials regarding the implementation of the response actions selected in the Draft ROD for the MI and to review the current regulatory framework in which final remedial goals may be developed for Dunn Field. The objectives of this section are to:

- Summarize and assess the findings and decisions of the ROD for the MI;
- Identify regulatory options to be considered during the 5-year ROD review for the MI; and
- Identify options and methodologies for developing final cleanup goals for Dunn Field.

10.1 FINDINGS AND DECISIONS OF THE DRAFT ROD FOR THE MAIN INSTALLATION

The Final ROD for the MI at DDMT (CH2M Hill, 2001b) establishes the selected remedial actions for soil and groundwater that will allow the MI property to be transferred for its intended use. The remedial action objectives (RAOs) and cleanup levels established in the ROD were selected based on the chemicals of concern (COCs) identified in the BRA report (CH2M Hill, 2000a), the proposed COC cleanup levels developed in the MI soil and groundwater FS reports (CH2M Hill, 2000e and 2000f), and considerations of current and reasonably anticipated future land uses. The findings of the BRA and FS reports are summarized in Section 2. The current and future land use assumptions and selected remedial alternatives presented in the Final ROD are discussed below.

10.1.1 Land Use Assumptions for the Main Installation

Per the Final ROD (CH2M Hill, 2001b), the overall strategy for remediating soils and groundwater at the MI was to select the most effective response actions that would allow transfer or lease of the property for its intended land use. The Memphis Depot Redevelopment Plan was approved by the Depot Redevelopment Corporation (DRC) board of directors, the City of Memphis, and Shelby County in 1997. According to the Redevelopment Plan, the intended land use for the MI is industrial for FUs 1, 3, 4, 5, and 6, and unlimited recreational for FU 2. In addition, the former Base housing area at FU 6 has been identified for use as transitional (temporary) housing for the homeless. The MI is zoned as Light Industrial (I-L), and the principal permitted land uses include manufacturing, wholesaling, or warehousing. According to Section 24 of the Memphis and Shelby County zoning regulation, single family, or multi-family residential uses are prohibited at the MI. Under the Federal Property Management Regulations, FU 2 is slated for transfer from the Department of Defense (DOD) to the Department of the Interior/National Park Service. The FU2 property will then be transferred by public benefit conveyance to the City of Memphis for use as a park. According to 41 CFR 101-47.308-7, property for use as a public park or recreational area must be used and maintained for the purpose for which it was conveyed in perpetuity, or be returned to the United States (24 CFR 51D).

Groundwater at the MI is classified as General Use Groundwater, as defined by the TDEC (1200-4-3-.07). General Use Groundwater is considered to be a potential source of drinking water, and chemical concentrations in these aquifers must be below drinking water criteria (i.e., MCLs). Groundwater at the Depot currently is not used for drinking water or other purposes, and is not likely to be used in the future (DRC, 1997). A well survey conducted within a 3-mile radius of the Depot did not identify any off-site residential or downgradient commercial wells pumping from the fluvial aquifer. In addition, groundwater use controls established by the Memphis-Shelby County Health Department, Water Quality Branch, prevent the installation of water wells within 0.5 mile of the designated boundaries of a listed Federal CERCLA site.

10.1.2 Selected Remedy for MI Soils

Per the Final ROD (CH2M Hill, 2001b), the selected remedy for MI soils was based on the following anticipated future land uses: unlimited recreational at FU 2, transitional residential at the housing area at FU 6, and industrial at FUs 1, 3, 4, 5, and the remainder of 6. As described in the soils FS (CH2M Hill, 2000e), FUs 1, 2, 3, 5, and 6 are suitable for their anticipated non-residential uses without any further action based on the interim soil removal actions that have already taken place at the MI. Therefore, the only remedial action necessary to address RAOs for surface soils at these FUs is to prevent residential use. At FU 4, lead concentrations above 1,536 mg/kg pose unacceptable risks to workers. Therefore, additional remedial action is required at FU 4 to ensure that the property is suitable for its intended future industrial use.

Based on these considerations, “institutional controls” (Alternative SS2) was selected as the remedial alternative for each FU (with variations among FUs), and “excavation and off-site disposal” (Alternative SS7) was selected as an additional remedy at FU 4. Alternative SS2 includes the use of deed restrictions to prevent residential use, including day care operations, at FUs 1, 3, 4, and 5. The same deed restrictions and site controls apply at FUs 2 and 6, but future unlimited recreational activities may occur at FU 2 and transitional residences may occur at the housing area at FU6. Alternative SS7 includes the excavation, transport, and off-site disposal of lead-contaminated surface soils at FU4.

Following excavation of the contaminated soil, one foot of clean backfill will be placed in all excavated areas, and the area's landscaping will be restored to its original condition. Alternative SS7 will require temporary controls at FU4 that will limit the use of those areas immediately adjacent to the excavation sites.

The established cleanup goal for lead in soils at FU 4 is 1,536 mg/kg (CH2M Hill, 2001a). Per the soils FS, this cleanup goal was developed based on guidance provided by USEPA (1996) in *Recommendations of the Technical Review Workgroup (TRW) for Lead for an Interim Approach to Assessing Risks Associated with Adult Exposures to Lead in Soil*. Using the TRW guidance and default adult industrial exposure parameters, estimated cleanup goals for lead in soil typically range from 700 to 1,700 mg/kg (USEPA, 1996). Therefore, the established cleanup level of 1,536 mg/kg for lead in soils at FU 4 is likely to be protective of future workers potentially exposed at the site.

Based on a review of the site investigation results summarized in Section 2, and the remedy-decision summary provided in the Draft ROD, the selected remedies for soils at the MI will likely achieve acceptable risk levels and allow for the anticipated land uses at the site.

10.1.3 Selected Remedy for MI Groundwater

Per the Final ROD (CH2M Hill, 2001b), the selected remedy for groundwater at the MI was chosen to be protective of human health and the environment and to meet the requirements of General Use Groundwater as defined by TDEC (12000-4-3-.07). Although groundwater from both the Fluvial aquifer and the underlying confined sand aquifer within the Jackson Formation/Upper Clairborne Group is classified as General Use Groundwater, groundwater at the Depot currently is not used as a drinking water source, and is not expected to be used as a drinking water source in the future (DRC, 1997).

Five groundwater remedial alternatives were identified and evaluated in the groundwater FS (CH2M Hill, 2000f): no action; MNA; enhanced bioremediation; air sparging; and extraction and discharge. Per the Final ROD, the preferred remedial alternative for groundwater is enhanced bioremediation. This alternative includes injection of nutrients/chemicals to enhance natural biodegradation processes, and institutional controls and groundwater monitoring. This alternative was selected to reduce potential human health risks within a reasonable time frame and to provide long-term reliability of the remedy.

MCLs for TCE (5 μg/L) and PCE (5 μg/L) were identified as applicable or relevant and appropriate requirements (ARARs) for groundwater beneath the MI based on the groundwater classification at the site. Per the Final ROD, the RAOs established for groundwater are expected to prevent ingestion of water from potential future on-site wells that contains contaminant concentrations in excess of MCLs; restore groundwater quality such that MCLs are not exceeded; and prevent offsite migration of groundwater contaminants at concentrations in excess of MCLs.

The Final ROD for the MI states that the selected remedy for groundwater is believed to represent a permanent solution. It should be noted that the long-term effectiveness and permanence of the selected remedy for groundwater may be limited if there is a significant continuing contaminant source in the subsurface at or upgradient of the Depot. The selected remedy is designed to treat the effects of the source (the dissolved plume), and not necessarily the source itself. Therefore, additional efforts to locate the source(s) are encouraged. The Responsiveness Summary in the ROD indicates that TDEC intends to conduct a site assessment of the potential off-site sources.

As described in Section 3, the Dunn Field site characterization activities performed in October 2000 provided a good test case for the use of the SimulProbe™ technology and allowed some definition of the way in which this technology can be used to advantage in field investigations. The experience at Dunn Field indicates that the SimulProbe™ or other similar device can be used successfully to identify the location of vadose zone VOC source areas. It can be an effective screening tool to determine if vadose zone impact exists within a general area (due to its apparent sensitivity). Once it is determined that soil gas impact is present, it is likely that the SimulProbe™ would be an effective tool for establishing the location of the impact.

Given that the SimulProbe™ appears to provide a stronger "signal" of VOC impact to soil gas than more conventional soil headspace screening, it can be inferred that this technique will be more likely to detect low-level contamination than headspace analysis. This inference is supported by the work of Siegrist and Jenssen (1990). This sensitivity could be valuable in screening an area for potential impact. An initial sampling location could be located a significant distance from a source area, and the presence of contamination could potentially still be detected in soil gas using the SimulProbe™, because the source area may be surrounded by a "cloud" of contaminated soil gas. Once soil gas impact was detected, a wide grid of borings could be advanced in the target area, and the SimulProbe™ could be used to move toward and eventually find the location of impact. Use of the SimulProbe™ in conjunction with a direct push device such as a cone penetrometer testing (CPT) rig (assuming that the rig could penetrate the subsurface to the desired depth) would be a cost-effective and low-profile source identification approach.

10.2 REGULATORY OPTIONS TO BE CONSIDERED DURING THE MI FIVE-YEAR REVIEW PROCESS

This section identifies regulatory options appropriate for consideration during the MI's five-year ROD review. As discussed in Section 10.1.2, the selected remedy for soils at the MI will likely achieve acceptable risk levels and allow the property to be transferred or leased for its intended land use. The achievement of MCLs in groundwater at the MI is less certain based on the potential limitations of existing remedial technologies and the lack of source identification. Therefore, the following discussion focuses on regulatory options available for remediation of groundwater at the MI.

10.2.1 USEPA's Technical-Impracticability Waiver

USEPA's (1993) technical-impracticability (TI) waiver protocol includes provisions for an exception to the application of MCLs as ARARs for sites where cleanup goals based on drinking water standards may not be achievable. ARARs may be waived by USEPA for any six of the reasons specified in the National Contingency Plan (under

CERCLA §121[d][4]), including TI from an engineering perspective. The TI evaluation generally should include the following components, based on site-specific information and analyses (USEPA, 1993):

- Specific ARARs or matrix-specific cleanup standards for which TI determinations are sought.
- Spatial area over which the TI waiver will apply.
- A CSM that describes site geology, hydrology, and groundwater contamination sources, transport, and fate.
- An evaluation of the restoration potential of the site, including data and analyses that support any assertion that attainment of ARARs or matrix cleanup standards is technically impractical from an engineering perspective, including as a minimum:
- A demonstration that contaminant sources have been identified and have been or will be removed and contained to the extent possible;
- An analysis of the performance of any ongoing or completed remedial action;
- Predictive analysis of the time frames to attain required cleanup levels using available technologies; and
- A demonstration that no other remedial technologies (conventional or innovative) could reliably, logically, or feasibly attain the cleanup levels at the site within a reasonable timeframe.
- Estimate of cost of the existing or proposed remedy options, including construction and OM&M costs.
- Any additional information or analyses that USEPA deems necessary for the TI evaluation.

If a TI decision is made, USEPA (1993) requires that an alternative cleanup strategy be developed. Site-specific cleanup goals may be developed for the affected media using USEPA's Risk-Based Corrective Action (RBCA) process as part of this alternate strategy. The RBCA process uses a tiered approach for site remediation (and the development of cleanup goals) where corrective action activities are tailored to site-specific conditions and risks. The potential application of the RBCA process at the MI is discussed in Section 10.2.3.

10.2.2 Tennessee State Rules for Groundwater Classification

TDEC (1200-4-3-.07) recognizes that not all impacted groundwater can be remediated. In these situations, the groundwater may be classified as "Site Specific Impaired" upon certification by the commissioner of Public Health. This groundwater classification means that drinking water MCLs do not necessarily have to be used as cleanup standards. Nonetheless, Site Specific Impaired groundwater cannot contain any toxic, carcinogenic, mutagenic, or teratogenic substances (other than background levels from natural origins) that pose an unreasonable risk to human health or the environment. The groundwater plume cannot extend off-site so that contaminants prevent groundwater beyond site boundaries from meeting their normal classification and criteria.

TDEC (1200-4-4-.09) describes the procedures used to apply for a Site Specific Impaired groundwater classification. Applications are evaluated based on the following criteria:

- The extent of any threat to human health or safety;
- The extent of damage to the environment;
- Availability of commercial technology to accomplish the restoration;
- A comparison of the environmental and economic costs and benefits to be derived from groundwater quality restoration with the environmental and economic costs and benefits to be derived from classification as Site Specific Impaired;
- The point of classification change; and
- Other appropriate information presented in the application.

If groundwater at the MI is declared "Site Specific Impaired," then an alternative cleanup strategy must be developed. The application of the RBCA process at the MI is discussed further in the following subsection.

10.2.3 Development of Alternative Groundwater Cleanup Goals

If a Site Specific Impaired groundwater classification is attained at the MI, then alternate groundwater cleanup goals will need to be developed. The RBCA process provides one method for calculating alternate site-specific cleanup goals. The RBCA process uses a tiered approach, which integrates site assessment and response actions with human health and ecological risk assessment, to evaluate the necessity for remedial action, and to tailor corrective actions to site-specific conditions and risks.

An alternate cleanup goal was calculated for TCE in groundwater to provide an example of the RBCA process in developing alternate groundwater cleanup goals. This alternate goal, which is compared to the MCL for TCE in Table 10.1, was developed based on a generic industrial land-use scenario and the following exposure assumptions:

- An onsite well will be used to supply water for landscape irrigation (no consumption);
- Groundskeepers may be exposed through dermal contact to contaminants in extracted groundwater that is used for landscape irrigation. (Exposures via the food ingestion chain were not evaluated in this example calculation because the COCs at DDMT are VOCs, which have a relatively low propensity to bioaccumulate. In

addition, inhalation exposures were not evaluated due to the dissipation of VOCs into ambient air);

- The hands, forearms, and lower legs would be the body parts dermally exposed based on assumed worker attire;

- A groundskeeper would irrigate an average of once per week for 50 weeks each year. The watering frequency in the warmer months may be 2 times a week, but this would be significantly less in the winter, averaging once a week for a given year; and

- The risk-based cleanup goal is based on a 1 in 1,000,000 (i.e., 1E-06) excess cancer risk.

Details of the exposure assumptions, models, and input parameters are presented in Appendix F.

TABLE 10.1
ALTERNATE RISK-BASED CLEANUP GOAL FOR TCE IN GROUNDWATER
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Groundwater COC	Alternate Risk-Based Cleanup Goal (µg/L)	MCL for Groundwater [a/] (µg/L)
TCE	1,260	5

[a/] Source: USEPA, 2000.

It should be noted that the risk-based cleanup goal for TCE presented in Table 10.1 is generic for industrial settings, and does not necessarily represent actual or expected exposure conditions at the industrial areas at the DDMT. Progress towards achieving MCLs for COCs in groundwater will be evaluated during the five-year ROD review at the MI. At that time, the appropriateness of establishing site-specific risk-based cleanup goals at DDMT should be considered.

10.3 DEVELOPMENT OF FINAL CLEANUP GOALS FOR DUNN FIELD

As discussed in Section 2, the nature and extent of potential contamination at Dunn Field has not been fully characterized. Previous investigations have shown that soils may be a continuing source of contamination to groundwater at Dunn Field (CH2M Hill, 2000b). Due to the ongoing investigations at Dunn Field, proposed RAOs and cleanup goals have not been developed for the site.

As described in Section 3.2, recommendations of this Phase II RPO evaluation include conducting SVE pilot testing to determine the feasibility and economics of operating a full-scale SVE system at Dunn Field. If upon completion of this pilot test, SVE is identified as the most effective long-term strategy for contaminant mass-removal in the vadose zone at Dunn Field, appropriate target soil gas concentrations and soil cleanup goals will need to be established. A recommended procedure for using a 1-D analytical model to derive soil cleanup goals that are protective of groundwater quality is presented in the following paragraphs.

10.3.1 Modeling Objectives

The purpose of site-specific, unsaturated-zone contaminant transport modeling for Dunn Field would be to evaluate the possible downward migration of COCs through the vadose zone to the water table, and to predict the maximum concentrations of these COCs that could remain in the vadose zone without allowing their continued migration to the water table at concentrations that would exceed groundwater cleanup levels (e.g., MCLs). The results of this evaluation can then be used to calculate the concentrations of COCs in the vapor phase, in equilibrium with the maximum sorbed and dissolved soil concentrations, that could remain in the soil column within the vadose zone.

These calculated vapor-phase concentrations of COCs would represent screening-level indicators of site-specific cleanup criteria for COCs in soil at Dunn Field. Compliance with soil cleanup goals could then be assessed using soil gas data collected from the VWs and VMPs installed as part of an SVE system. If vapor-phase concentrations of COCs exceed the screening-level soil-vapor cleanup criteria, then it is likely that the concentrations of these COCs in the sorbed, dissolved, and/or vapor phases in the vadose zone are sufficiently elevated that the COCs will continue to migrate to the water table at concentrations that would exceed the groundwater cleanup goals. Conversely, if vapor-phase concentrations are below the screening-level soil-vapor cleanup criteria, then migration of these compounds to the water table at concentrations that would exceed the cleanup goals is unlikely to occur.

10.3.2 Modeling Approach

Water percolation through the soil column can be computed using the Hydrologic Evaluation of Landfill Performance (HELP) code (Schroeder *et al*., 1994). Site-specific soil and climatic data are incorporated into the model in which vertical, unsaturated movement of liquid water is assumed to occur primarily due to gravitational forces. The percolation out of the bottom layer of the soil column simulated in the HELP model represents the amount of liquid water flux (recharge) to the groundwater system. The HELP model results would then be used as input into the 1-D vadose zone contaminant transport model described below.

Use of an analytical solution to the 1-D, unsaturated, contaminant-mass transport equation described by Jury *et al*., (1983) is recommended to evaluate the potential migration of COCs in the subsurface at Dunn Field. Using the "Jury" model, chemical migration in the aqueous phase can be examined, and because the vadose zone contains some proportion of air in the pore spaces, vapor-phase transport also is accounted for. The solution to the equations describing 1-D, unsaturated transport (Jury *et al*., 1983) is in the form of a partitioning model that distributes a chemical species in equilibrium among its possible phases (dissolved in water, sorbed to soil, and in soil vapor) in accordance with its chemical properties and local conditions in the subsurface. An advantage of using the Jury model is that the vapor-phase chemical concentration in equilibrium with the sorbed and dissolved chemical phases can be calculated, and the vapor-phase flux at the ground surface can be calculated for any point in time.

Use of a 1-D analytical solution of this type is appropriate for the following reasons:

- The analytical solution considers the effects of the principal physical and chemical mechanisms that contribute to chemical migration and environmental attenuation: advection, diffusion, dispersion, volatilization, sorption, decay, and source-mass depletion.

- Use of 2-D or 3-D variably-saturated flow and transport codes (e.g., VLEACH or SESOIL) requires that the subsurface be characterized in considerable detail, in order to provide the quality and quantity of input data required for accurate numerical modeling. The required level of detail generally is not available.

- Because a conservative, site-specific, average infiltration rate can be estimated, and because accurate modeling of transient infiltration is problematic, it may be appropriate to disregard numerical simulations of flow, and instead use a calculation method that requires only steady-state seepage velocity as a primary input.

- Analytical solutions are simpler (and less expensive) to implement, have fewer sources of error, and are easier to verify than comparable numerical solutions. Their use is preferable in cases where available data are insufficient to take advantage of a numerical solution's greater flexibility.

Subsurface transport of chemicals as NAPL or in the aqueous or vapor phase is driven by potential gravitational, hydraulic, and/or chemical gradients. In the unsaturated zone, gravitational and hydraulic potential gradients are primarily vertical, so that the direction of movement of chemicals in the dissolved or NAPL phases is generally downward. The atmosphere represents the ultimate sink for VOCs, so that vapor-phase chemical concentration gradients are usually directed upward, and vapor-phase migration is induced from the subsurface to the atmosphere. As a consequence of the generally vertical orientation of gravitational, hydraulic, and chemical gradients, application of a 1-D solution to the evaluation of conditions in the vadose zone is entirely appropriate, because dissolved-phase chemical migration in a 1-D model occurs in only one direction (downward). In examining precipitation, infiltration, and chemical flux through the vadose zone, water and the various chemical species (sorbed, aqueous, and vapor) are assumed to originate at some interval below the ground surface, and to travel straight downward through the soil column.

In summary, a 1-D analytical model simplifies many of the complexities of a "real" vadose-zone transport system. Nevertheless, if the available data are limited and the transport parameters are suitably restricted, a 1-D model can provide sufficient information for a first-order assessment of chemical migration and possible environmental impacts (Javandel *et al.*, 1984).

Prior to applying the Jury model to calculation of residual concentrations of PCE and TCE sorbed to soil and in soil vapor within the vadose zone that would be protective of groundwater quality, the model should be used to examine the vertical distribution of contaminants in the vadose zone at a site on Dunn Field to evaluate whether the model structure is valid, and whether the values of input parameters to be used in subsequent simulations are representative of actual conditions. If model-predicted and measured contaminant concentrations do not compare well, then model input parameters, such as the constant describing the rate of chemical degradation ("first-order rate constant") can be regarded as model calibration parameters, and can be adjusted until the calculated distribution of the modeled compound is in reasonable agreement with the observed vertical distribution of the compound in the soil column. At this point, the model should be regarded as valid, and was used in subsequent simulations to calculate the concentrations of COCs in soil vapor that would be protective of groundwater quality.

SECTION 11

CTC AND STC

CTC and STC estimates for DDMT are provided in this section. Parsons ES was unable to access the Critical Path Planning Toolbox (CPPT) from the AFCEE web page, and therefore was unable to integrate the CTC and STC with the CPPT as described in the work plan (Parsons ES, 2000b).

11.1 COST-TO-COMPLETE

The estimated cost to complete remediation for DDMT is approximately $20.1 million. This estimate is summarized in Table 11.1, and includes estimated costs for fiscal year (FY) 2001 and beyond for anticipated completion of remediation activities. The CTC in Table 11.1 is summarized on a total cost basis (i.e., not discounted for present worth). Supporting cost tables that provide additional details are contained in Appendix E.

Following is a discussion of the assumptions associated with the CTC for the MI and Dunn Field.

11.1.1 Main Installation

The CTC for the MI in Table 11.1 is subdivided into source area costs and groundwater remediation costs. The long-term success of the proposed groundwater remedy, which is enhanced bioremediation as described in the ROD (Memphis Depot Caretaker Division, 2001), is highly dependent on remediation of source areas and application of the *in situ* treatment to the most significantly-impacted portion of the aquifer. It is anticipated that additional delineation of source areas will be required to optimize the proposed remedy. Therefore, additional site characterization costs have been included in FY 2002 to delineate source areas that contribute to groundwater contamination. This additional site characterization could also be considered as pre-design investigations to optimize the selected remedy.

Capital costs for soils remediation are the same as described in the ROD, and were assigned to be incurred in FY 2001. Capital costs as well as initial O&M costs for groundwater remediation are generally the same as described for enhanced bioremediation in the ROD, and are expected to be incurred in FY 2003 with O&M costs through 2016. Yearly O&M costs for enhanced bioremediation are the same as described in the ROD. Costs for semiannual and annual groundwater monitoring are expected to be incurred through 2021 based on successful bioremediation treatment through 2016 (see Section 6.3), plus an additional 5 years for verification of attainment of cleanup goals.

TABLE 11.1
COST-TO-COMPLETE ESTIMATE
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

General Area/ Operable Unit	Total Cost	Line Item	Subtotal Item Cost	FY 2001	FY 2002	FY 2003	FY 2004	FY 2005	FY 2006	FY 2007	FY 2008
Main Installation -											
Source Area	$4,155,000	Site characterization/design	$852,000		$500,000	$352,000					
(see Table E1)		Reporting and regulatory support	$0								
		Capital costs	$849,900	$183,000		$666,900					
		Operation and maintenance	$2,453,100				$188,700	$188,700	$188,700	$188,700	$188,700
		Remedial system monitoring	$0								
		Long-term monitoring	$0								
		Major upgrades	$0								
Main Installation -											
Groundwater	$999,947	Site characterization	$0								
(see Table E2)		Reporting and regulatory support	$172,800	$10,560	$10,560	$10,560	$10,560	$10,560	$40,000		
		Capital costs	$0			$0					
		Operation and maintenance	$0								
		Remedial system monitoring	$0								
		Long-term monitoring	$786,647			$74,919	$74,919	$74,919	$74,919	$74,919	$37,459
		Major upgrades	$40,500								
Dunn Field -											
Source Areas and											
Groundwater	$14,482,049	Site characterization	$350,000	$350,000							
(see Tables E3		Reporting and regulatory support	$585,000		$135,000	$30,000	$30,000	$30,000	$30,000	$30,000	$70,000
and E4)		Capital costs	$2,404,156	$1,290,956		$1,113,200					
		Operation and maintenance	$7,189,998	$167,236	$167,236	$167,236	$300,796	$300,796	$300,796	$300,796	$300,796
		SVE performance monitoring	$211,200				$42,240	$42,240	$42,240	$42,240	$42,240
		Long-term monitoring	$2,716,284	$118,099	$118,099	$118,099	$118,099	$118,099	$118,099	$59,050	$59,050
		Major upgrades	$1,025,411			$128,176					$128,176
TOTAL COST:	$19,636,997	Subtotals:	$19,636,997	$2,119,851	$930,895	$2,661,090	$765,314	$765,314	$794,754	$695,704	$826,421

Note: Tables E1 - E4 are in Appendix E

TABLE 11.1 (Continued)
COST-TO-COMPLETE ESTIMATE
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

General Area/ Operable Unit	Total Cost	Line Item	FY 2009	FY 2010	FY 2011	FY 2012	FY 2013	FY 2014	FY 2015	FY 2016	FY 2017	FY 2018	FY 2019
Main Installation -													
Source Area	$4,155,000	Site characterization/design											
(see Table E1)		Reporting and regulatory support											
		Capital costs											
		Operation and maintenance	$188,700	$188,700	$188,700	$188,700	$188,700	$188,700	$188,700	$188,700			
		Remedial system monitoring											
		Long-term monitoring											
		Major upgrades											
Main Installation -													
Groundwater	$999,947	Site characterization											
(see Table E2)		Reporting and regulatory support			$40,000					$40,000			
		Capital costs											
		Operation and maintenance											
		Remedial system monitoring											
		Long-term monitoring	$37,459	$37,459	$37,459	$37,459	$37,459	$37,459	$37,459	$37,459	$37,459	$37,459	
		Major upgrades						$40,500					
Dunn Field -													
Source Areas and													
Groundwater	$14,482,049	Site characterization											
(see Tables E3		Reporting and regulatory support	$30,000				$40,000					$40,000	
and E4)		Capital costs											
		Operation and maintenance	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236
		SVE performance monitoring											
		Long-term monitoring	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050
		Major upgrades					$128,176					$128,176	
TOTAL COST:	$19,636,997	Subtotals:	$482,445	$452,445	$492,445	$452,445	$620,621	$492,945	$452,445	$492,445	$263,745	$431,921	$226,285

Note: Tables E1 - E4 are in Appendix E

TABLE 11.1 (Continued)
COST-TO-COMPLETE ESTIMATE
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

General Area/ Operable Unit	Total Cost	Line Item	FY 2020	FY 2021	FY 2022	FY 2023	FY 2024	FY 2025	FY 2026	FY 2027	FY 2028	FY 2029	FY 2030
Main Installation - Source Area (see Table E1)	$4,155,000	Site characterization/design											
		Reporting and regulatory support											
		Capital costs											
		Operation and maintenance											
		Remedial system monitoring											
		Long-term monitoring											
		Major upgrades											
Main Installation - Groundwater (see Table E2)	$999,947	Site characterization											
		Reporting and regulatory support											
		Capital costs											
		Operation and maintenance											
		Remedial system monitoring											
		Long-term monitoring											
		Major upgrades											
Dunn Field - Source Areas and Groundwater (see Tables E3 and E4)	$14,482,049	Site characterization											
		Reporting and regulatory support				$40,000					$40,000		
		Capital costs											
		Operation and maintenance	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236
		SVE performance monitoring											
		Long-term monitoring	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050
		Major upgrades				$128,176					$128,176		
TOTAL COST:	$19,636,997	Subtotals:	$226,285	$226,285	$226,285	$394,462	$226,285	$226,285	$226,285	$226,285	$394,462	$226,285	$226,285

Note: Tables E1 - E4 are in Appendix E

TABLE 11.1 (Continued)
COST-TO-COMPLETE ESTIMATE
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

General Area/ Operable Unit	Total Cost	Line Item	FY 2031	FY 2032	FY 2033	FY 2034	FY 2035	FY 2036	FY 2037	FY 2038	FY 2039	FY 2040
Main Installation -												
Source Area	$4,155,000	Site characterization/design										
(see Table E1)		Reporting and regulatory support										
		Capital costs										
		Operation and maintenance										
		Remedial system monitoring										
		Long-term monitoring										
		Major upgrades										
Main Installation -												
Groundwater	$999,947	Site characterization										
(see Table E2)		Reporting and regulatory support										
		Capital costs										
		Operation and maintenance										
		Remedial system monitoring										
		Long-term monitoring										
		Major upgrades										
Dunn Field -												
Source Areas and												
Groundwater	$14,482,049	Site characterization										
(see Tables E3		Reporting and regulatory support			$40,000							
and E4)		Capital costs										
		Operation and maintenance	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	$167,236	
		SVE performance monitoring										
		Long-term monitoring	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050	$59,050
		Major upgrades			$128,176					$128,176		
TOTAL COST:	$19,636,997	Subtotals:	$226,285	$226,285	$394,462	$226,285	$226,285	$226,285	$226,285	$354,462	$226,285	$59,050

Note: Tables E1 - E4 are in Appendix E

Additional costs associated with groundwater remediation (independent of source area treatment) include regulatory support by attending Restoration Advisory Board meetings for an additional five years.

11.1.2 Dunn Field

Backup information for source area and groundwater remediation costs for Dunn Field is provided in Appendix E. SVE has been identified as a potentially viable technology for source area remediation to eliminate the vadose zone sources of groundwater contamination. SVE treatment of source area soils is expected to allow the groundwater extraction system to cease operation after attainment of groundwater cleanup goals, which is predicted to occur by 2039 (Section 5.3).

Additional site characterization costs have been included in FY 2001 to complete the delineation of potential source areas at Dunn Field. Completion of an FS, Proposed Plan, and ROD is anticipated to be completed by FY 2002, followed by design and implementation of a source area SVE system in 2003. The SVE systems are assumed to operate for five years, which is expected to be sufficient to remove available vadose zone VOCs. Additional groundwater monitoring immediately downgradient of the SVE wells would be performed to assess the performance of the remediation systems.

Operation and maintenance costs include the continued operation of the existing groundwater extraction system with direct discharge of untreated water to the Memphis sanitary sewer system. SVE O&M costs are also included from FY 2004 through FY 2008. Long-term monitoring costs are based on current actual costs for quarterly monitoring through FY 2006, which is assumed to be reduced to semiannual monitoring after two years of source-area remediation. Periodic repairs and upgrades to the groundwater extraction system are anticipated to occur every five years.

Estimated O&M of the extraction system comprises the majority of the total cost for Dunn Field. Although a more aggressive groundwater remediation strategy (such as enhanced bioremediation) may result in a shorter duration, this may not be appropriate for Dunn Field because of the proximity of the plume to the Base boundary and the potential for generation and offsite migration of VC (Section 6).

11.2 SCHEDULE-TO-COMPLETE

The CTC is significantly impacted by the anticipated schedule-to-complete (STC) for groundwater remediation, which is provided in Figure 11.1. The STC is based on an approximation of contaminant decay in the groundwater, as described in Section 5 of this report. Assuming removal of source area contributions (i.e., source area remediation), TCE concentrations at Dunn Field are predicted to decay to the 5-μg/L MCL by the year 2039. Based on predictions outlined in this report (Section 6.3), PCE concentrations in Main Installation groundwater are predicted to decay to below the 5 μg/L MCL by the year 2016. If no enhanced treatment options were exercised, then PCE concentrations are not predicted to fall below MCLs for at least 30 years. These calculations are sensitive to site-specific input parameters that may not be well-defined (e.g., the organic carbon content in the aquifer), so the actual STC may vary significantly from these predictions.

Figure 11.1 Schedule-to-Complete

SECTION 12

RECOMMENDATIONS AND IMPLEMENTATION PLAN

The following recommendations are made based on the results of the RPO Phase II evaluation:

Recommendation No. 1: A SVE pilot test should be performed at Dunn Field as described in Section 3. Prior to performance of the pilot test, two additional VMPs should be installed as described in Section 3.2.3. In addition, an existing monitoring well should be identified for temporary use as a VMP. If a full-scale SVE system is implemented, then the decision tree presented in Section 4.3 should be used to optimize system operation over time and justify eventual system shutdown.

<u>Rationale:</u> *Much of the pilot test infrastructure was installed in October 2000, and drilling/installation of the recommended additional VMPs could be readily performed. Performance of the pilot test will facilitate realistic evaluation of this remedial alternative during the upcoming FS, and accelerate site cleanup (i.e., a likely recommendation of the FS would be to perform an SVE pilot test to determine mass removal rates, radius of influence, etc.).*

<u>Implementation:</u> *This recommendation should be implemented ASAP to support the remedial decision-making process (i.e., the FS for soils). This draft report should first be reviewed, and the required consensus on the recommendations obtained. The recommendations described in Section 3 should then be used to prepare a detailed pilot test work plan within four months of acceptance of this recommendation. Once the work plan is finalized, the SVE pilot test should be performed. Installation of the two to three recommended additional VMPs could be performed any time prior to performance of the pilot test. Completion of the pilot test (including data analysis/interpretation and reporting) within 5 months of work plan finalization (i.e., within 9 months of acceptance of this recommendation) should be targeted to support the FS.*

Recommendation No. 2: Vadose zone cleanup goals for Dunn Field should be developed that are based on the relationship between average equilibrium soil gas concentrations in the soil column and the groundwater protection or risk-based soil standard for the site. A procedure for deriving these cleanup goals is described in Section 10.3.

Rationale: *Compliance with soil cleanup goals could be assessed using soil gas data collected from the VWs and VMPs installed as part of an SVE system. If vapor-phase concentrations of COCs exceed the screening-level soil-vapor cleanup criteria, then it is likely that the concentrations of these COCs in the sorbed, dissolved, and/or vapor phases in the vadose zone are sufficiently elevated that the COCs will continue to migrate to the groundwater table at concentrations that would exceed the groundwater cleanup goals. Conversely, if COC vapor-phase concentrations are below the screening-level soil-vapor cleanup criteria, then migration of these compounds to the water table at concentrations that would exceed the cleanup goals is unlikely to occur. Assessing compliance with cleanup goals via drilling and soil sampling is costly and problematic given the frequent heterogeneity of contaminant distribution in subsurface soils and the problems associated with obtaining representative VOC concentrations in sandy soils.*

Implementation: *Implementation of this recommendation prior to issuance of the SVE pilot test results report would allow upfront assessment of the degree to which soil gas concentrations exceed recommended cleanup criteria, thereby facilitating evaluation of the anticipated time frame to achieve cleanup. Measured VOC concentrations in soil gas, obtained during pilot testing activities, should be compared to recommended cleanup criteria in the pilot test results report. This report would then comprise a complete "package" that would be of maximum benefit during the remedial decision-making process.*

Recommendation No. 3: If enhanced bioremediation of CVOCs in groundwater beneath the southwestern portion of the MI is performed (as described in the ROD), then a vegetable oil pilot test should be performed instead of, or in addition to, a pilot test utilizing the Regenesis product HRC®.

Rationale: *The current cost of vegetable oil is one to two orders-of-magnitude lower than the cost of HRC® listed in the draft ROD for the MI (CH2M Hill, 2001a). In addition, vegetable oil has a lower solubility than HRC, and will likely act as a slow-release carbon source for several years, eliminating the need for annual reinjection. Use of vegetable oil at several sites across the country has shown that it can effectively promote reductive dehalogenation of highly chlorinated solvents such as PCE and TCE.*

Implementation: *Assuming that the draft ROD for the MI is signed, this recommendation should be implemented in conjunction with source-area characterization (see Recommendation No. 7) to support location of the pilot test wells and design of an optimal full-scale enhanced bioremediation system. The pilot test should be performed over a 1- to 2-year period to allow a realistic assessment of the effectiveness of this technology and to facilitate a more accurate prediction of the time required to achieve MCLs in groundwater.*

Recommendation No. 4: Delete four monitoring wells (MW30, MW40, MW58, and MW59) from the current Dunn Field groundwater monitoring program, and adjust sampling frequencies from quarterly to semiannual to annual as specified in Section 8. This recommendation is preliminary in that is based on analytical results for TCE only; the recommendation should be reevaluated using recently collected hydrogeologic and chemical data, data for other COCs, and the process described in Sections 4.1 and 8, and then implemented as appropriate. The evaluation process described in Sections 4 and 8 should be used to design an optimized LTM program for both Dunn Field and the MI after the final remedies have been implemented. The most recent Dunn Field Groundwater monitoring data should be reviewed and deletion of SVOCs and pesticides from the target analyte list for Dunn Field groundwater monitoring should be considered, as described in Section 8.3.

Rationale: *Based on historical groundwater quality data for well MW30, and the cross-gradient location of this well, it is reasonable to conclude that TCE at that location will remain below detection limits in the future. Well MW40 is located approximately 800 to 900 feet cross-gradient from the boundary of the plume. This well is recommended for exclusion from the monitoring network because it is not particularly useful for defining the lateral boundary of the plume or for detecting potential future migration of contaminants. Wells MW58 and MW59 are recommended for exclusion from the monitoring network because they appear to be redundant with wells MW56 and MW68, respectively. Quarterly sampling of each newly-installed well for one year is recommended to establish baseline water quality conditions. Based on the moderate groundwater migration rates computed for Dunn Field, dramatic changes in contaminant concentrations at a particular location due to plume movement are not expected to occur very rapidly. Semiannual to annual sampling frequencies should be adequate to identify potential changes in concentrations that may require an action. Recent groundwater monitoring data for Dunn Field indicate that pesticides, and possibly SVOCs, are not groundwater COCs in the monitored portion of Dunn Field.*

Implementation: *Prior to implementation of this recommendation, it should be reviewed in light of the most recent groundwater monitoring data available to ensure that it is not contradicted by the data. In addition, data for other COCs besides TCE should be assessed as described in Section 8. If the most recent monitoring data and data for other COCs do not nullify this recommendation, then it should be reviewed with the regulators at an upcoming Restoration Advisory Board (RAB) meeting to obtain their concurrence. The sampling frequency adjustments recommended in Table 8.1 should be implemented as soon as possible.*

Recommendation No. 5: The f_{oc} content of the Fluvial aquifer should be clarified, and the contaminant decay rate calculations and groundwater modeling predictions described in Section 5 and 7, respectively, should be revisited and revised as appropriate if the representative f_{oc} value is significantly different from that used in the calculations.

Rationale: *Calculated decay rates for COCs (e.g., PCE and TCE) are used to estimate the time necessary to achieve cleanup goals in groundwater. The decay rate calculations presented in Section 5 are very sensitive to the magnitude of this parameter, and the value assigned to this parameter in various historical documents has varied. The migration predictions for TCE and PCE in MI groundwater, described in Section 7.1, also are affected by the magnitude of f_{oc}. Therefore, clarification of this value would allow greater confidence in the predicted cleanup time frames and plume migration distances.*

Implementation: *It is possible that this recommendation could be implemented by reviewing the existing database of soil f_{oc} values, and comparing the database to average values specified for this parameter in RI/FS reports. If the database is insufficient to confidently characterize the average f_{oc} value for the Fluvial aquifer, then additional f_{oc} data should be collected during any future scheduled drilling (e.g., drilling performed for the SVE or enhanced bioremediation pilot tests). Collection of soil samples for f_{oc} analysis from below the water table in the Fluvial aquifer is preferred; however, data collected from similar lithologies in the vadose zone near the water table also could be used.*

Recommendation No. 6: Vertical profiling of contaminant concentrations using PDB samplers should be performed in all monitoring wells that are sampled on a regular basis for VOCs only, as described in Section 9.4. The results should be compared to conventional sampling results in a scientifically defensible manner using appropriate statistical analyses to ascertain the representativeness of the PDB data and desirability of replacing conventional sampling with PDB sampling. Potentially appropriate statistical tests include the following:

- Analysis of variance (ANOVA) and/or two-sided approximate t-tests to compare the means of different groups of data to determine if there are statistical differences among the groups.
- Use of two-sided, paired t-tests to determine if the results of diffusion sampling are accurate and representative of groundwater concentrations on an installation-wide basis.
- Use of the Wilcoxen Signed-Rank Test to determine if there is any systematic (i.e., installation-wide) differences between the two data sets based on well depth or changes in lithology.
- Calculation of relative percent differences on a well-by-well basis to assess the equivalence of the diffusion sample and conventional sample data sets.

Rationale: *Assuming that PDB samplers for one sampling event can be deployed at the same time as the prior event's samplers are recovered, a savings of approximately $160 per sample per event could be realized. This translates into an annual per-well savings of $640, $320, and $160 for wells monitoring quarterly, semiannually, and annually, respectively. If*

40 wells are sampled for VOCs over a 30-year period (half semiannually and half annually), then a savings of approximately $290,000 could be realized.

Implementation: *Implementation of PDB sampling in the near future to the extent feasible would allow the maximum long-term cost savings to be realized. For this reason, the recommended vertical profiling and comparison with conventional sampling should be performed ASAP in the portion of Dunn Field that is currently monitored on a regular basis if the sampling frequencies indicate that significant cost-savings would be achieved. Selection of MI wells for vertical profiling prior to development of final LTM plans for groundwater runs the risk that wells will be profiled that will later be excluded from the LTM program. Therefore, this recommendation should be implemented at the MI once the scope of the final LTM programs (e.g., numbers and locations of wells, analyte list, and sampling frequencies) are determined. In this way, a more accurate assessment of potential cost savings and the desirability of PDB sampling can be determined. In addition, results of an upcoming, extensive, AFCEE initiative to compare PDB sampling results to conventional results will be available in 2002. These results could assist the Depot in gaining regulatory approval for implementation of PDB sampling.*

Recommendation No. 7: Additional efforts to locate the source(s) of the CVOC contamination in groundwater beneath the southwestern portion of the MI should be made using the SimulProbe™ or similar device.

Rationale: *The selected remedy for MI groundwater is designed to treat the effects of the source (the dissolved plume), and not necessarily the source itself. Without source treatment, dissolved COC concentrations in excess of cleanup levels may persist in MI groundwater indefinitely. The experience at Dunn Field indicates that the SimulProbe™ or other similar device can be used successfully to identify the location of vadose zone VOC source areas. It can be an effective screening tool to determine if vadose zone impacts exist within a general area (due to its apparent sensitivity). Once it is determined that soil gas impact is present, it is likely that the SimulProbe™ would be an effective tool for establishing the location of the impact.*

Implementation: *This recommendation should be implemented ASAP to support optimal location of enhanced bioremediation pilot test wells. The draft ROD for the MI (CH2M Hill, 2001a) indicated that TDEC is considering performing source location activities southwest of (hydraulically upgradient from) the Depot. Therefore, this recommendation should be discussed with TDEC at an upcoming RAB meeting.*

Recommendation No. 8: If the source(s) of the CVOC plumes beneath the southwestern portion of the MI cannot be found, and if future groundwater monitoring indicates that dissolved CVOC concentrations both on and upgradient from the MI are not decreasing at an acceptable rate, then classification of MI groundwater as Site Specific Impaired should be sought. If granted, alternate, risk-based cleanup goals should be developed as described in Section 10.2.3.

Rationale: *TDEC recognizes that not all impacted groundwater can be remediated. In these situations, the groundwater may be classified as "Site Specific Impaired" upon certification by the commissioner of Public Health. This groundwater classification means that drinking water MCLs do not necessarily have to be used as cleanup standards. Development of alternate, risk-based cleanup levels (Section 10.2.3) that are appropriate for the selected land use and receptors (e.g., industrial worker) may eliminate the need for engineered cleanup of groundwater.*

Implementation: *This recommendation should be considered as the 5-year review of the MI ROD approaches. Available data should be reviewed 1 year prior to the 5-year review to allow assessment of progress toward source characterization and achievement of groundwater cleanup goals.*

The suggested implementation schedule for the RPO recommendations listed above is summarized in Table 12.1.

TABLE 12.1
RECOMMENDED IMPLEMENTATION SCHEDULE
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

Recommendation No.	Item	Timeframe	Schedule
General	Review of Draft RPO Phase II report and consensus on recommendations	To be completed 3 weeks after delivery of draft report	2/19/01-3/9/01
General	Meeting with regulatory agencies	Within 1 month of completion of draft report review	3/10/01-4/10/01
1	SVE pilot test work plan	Within 4 months of meeting with regulatory agencies	4/10/01-8/10/01
1	SVE pilot test, data analysis, and results report	Within 5 months of work plan finalization	8/10/01-1/10/02
2	Development of vadose zone cleanup levels for CVOCs at Dunn Field	This effort should be part of the SVE pilot test data analysis and reporting task	By 1/10/02
3	Enhanced bioremediation pilot test	Perform in conjunction with source characterization activities at MI (Recommendation No. 7)	9/1/01-9/1/03
4	Finalization of current Dunn Field monitoring program evaluation and decision to exclude wells and revise analyte list	This recommendation should be implemented within 6 months of completion of draft report review	By 9/10/01
5	Evaluation of f_{oc} content of Fluvial aquifer	Review of the f_{oc} database could be performed within 2 months of completion of draft report review. If additional f_{oc} data are required, then data collection can be scoped into future drilling efforts.	By 5/10/01
6	Vertical profiling using PDB samplers	Initiate at Dunn Field following completion of draft report review; phase into MI program as appropriate.	Start 8/10/01 (Dunn Field)
7	MI CVOC source location/characterization	Source location activities should be performed within the 1-year period following completion of draft report review in order to support the selected remedy for MI groundwater	3/10/01-3/10/02
8	Consideration of Site Specific Impaired classification for MI groundwater	This assessment should be performed within the 1-year period prior to the 1st 5-year review of the MI ROD, assumed to occur in January 2003	1/02-1/03

SECTION 13

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APPENDIX A

FIELD SCREENING AND LABORATORY ANALYTICAL DATA AND SVE WELL CONSTRUCTION DIAGRAMS

APPENDIX B

BIODEGRADATION RATE CALCULATIONS

APPENDIX C

SUPPORTING DATA FOR MAIN INSTALLATION PLUME STABILITY EVALUATION

APPENDIX D

SUPPORTING DATA FOR DUNN FIELD GROUNDWATER MONITORING PROGRAM EVALUATION

APPENDIX E

SUPPORTING DATA FOR CTC ESTIMATE

TABLE E.1
BACKUP COSTS FOR COST-TO-COMPLETE ESTIMATE (MAIN INSTALLATION SOURCE AREA)
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

CAPITAL COSTS	Quantity	Units	Unit Rate	Summary Cost	
Enhanced bioremediation of groundwater; costs as described under Alternative GW3 in the MI Record of Decision (ROD) (less cost for pilot study, see below).	1	LS	$666,900	$666,900	
Soil excavation/removal and institutional controls; costs as described under Table 2-13 in the Record of Decision.	1	LS	$183,000	$183,000	
			TOTAL CAPITAL COSTS:	$849,900	

OPERATION AND MAINTENANCE (O&M) COSTS

Assume operation of enhanced bioremediation system until 2017 to reduce source area contributions.
Cost based on estimate for preferred alternative GW3 as described in the ROD.

	Quantity	Units	Unit Rate	Summary Cost	
O&M Costs (less monitoring costs from Table E.2)	1	year	$184,900	$184,900	per year
Annual evaluations and 5-year reviews from ROD Table 2-13.	1	year	$3,800	$3,800	per year
SUMMARY TOTAL O&M COSTS:				$188,700	per year

REMEDIAL SYSTEM PERFORMANCE MONITORING COSTS
Included in O&M costs, as described under Alternative GW3 in the ROD.

MONITORING WELL NETWORK/SURFACE WATER/AIR SAMPLING COSTS (covered under groundwater costs)

MAJOR UPGRADE COSTS (none anticipated)

FUTURE SITE CHARACTERIZATION

Detailed vadose zone and groundwater investigation to define source areas; Simulprobe with direct push sampling, followed by HAS drilling at hot spots. Cost based on engineering judgement.

$500,000 lump sum

FUTURE REPORTING AND REGULATORY SUPPORT

Future regulatory support (e.g., regulatory support, RAB meeting attendance, etc.) - Covered under groundwater costs.

OTHER COSTS

	Quantity	Units	Unit Rate	Summary Cost
Enhanced bioremediation pilot study/design (from ROD)	1	LS	$352,000	$352,000

TABLE E.2
BACKUP COSTS FOR COST-TO-COMPLETE ESTIMATE (MAIN INSTALLATION GROUNDWATER)
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

CAPITAL COSTS	Quantity	Units	Unit Rate	Summary Cost	
Covered under "Main Installation Source Area".	1	LS	$0	$0	
TOTAL CAPITAL COSTS:				$0	
OPERATION AND MAINTENANCE (O&M) COSTS - Covered under "Main Installation Source Area"					
MONITORING WELL NETWORK/SURFACE WATER/AIR SAMPLING COSTS Costs same as Table 2-14 of ROD. Assmues semiannual sampling for first 5 years, followed by annual sampling. Costs below are for annual sampling (use twice these costs for semiannual sampling).					
Analytical costs (VOCs and MNA monitoring)	23	each	$648	$14,904	per year
Sampling Labor and ODCs					
Mob/demob, use 32 hrs. X $75/hr = $2400 per event	1	each	$2,400	$2,400	per year
Sample labor	140	each	$75	$10,500	per year
Equip. and supplies, use $2800 per event	1	each	$2,800	$2,800	per year
Labor (data validation and evaluation)	46	hours	$75	$3,450	per year
Project management/reporting (10% of monitoring costs)	0.1	percent	$34,054	$3,405	per year
TOTAL MONITORING COSTS:				$37,459	per year
MAJOR UPGRADE COSTS				Summary Cost	Estimated Completion Date
Well abandonment (cost from the MI Groundwater FS Report)	45	ea	$900	$40,500	Following treatment
TOTAL MAJOR UPGRADE COSTS:				$40,500	
FUTURE SITE CHARACTERIZATION					
None anticipated.				$0	lump sum
FUTURE REPORTING AND REGULATORY SUPPORT					
5-year reviews				$40,000	per every 5 years
Future regulatory support (e.g., regulatory support, RAB meeting attendance, etc.)	96	hours	$110	$10,560	per year
OTHER COSTS					
None anticipated.					

TABLE E.3
BACKUP COSTS FOR COST-TO-COMPLETE ESTIMATE (DUNN FIELD SOURCE AREA)
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

CAPITAL COSTS	Quantity	Units	Unit Rate	Summary Cost		
Multiple-well vapor extraction system to reduce source area contributions to groundwater contamination. Unit rate based on App. D costs in the Draft Final Main Installation (MI) Groundwater FS Report; also similar to Federal Remediation Technologies Roundtable (FRTR) case history costs.	12	SVE wells	$75,000	$900,000		
Install vapor treatment system, unit rate based on engineering judgement. Assumes two systems required due to area of site.	1	LS	$200,000	$200,000		
Install additional groundwater monitoring wells for performance monitoring of SVE system. Unit rate based on App. D costs in the Draft Final Main Installation (MI) Groundwater FS Re	6	wells	$2,200	$13,200		
TOTAL CAPITAL COSTS:				$1,113,200		
OPERATION AND MAINTENANCE (O&M) COSTS				Detailed Cost		
Assume operation for five years to reduce source area contributions.						
Labor (SVE and plant operations); 0.5 FTE	1040	hours	$75	$78,000	per	year
Utilities (electricity, natural gas)	12	month	$2,000	$24,000	per	year
Consumable supplies and chemicals (include carbon regeneration, acids and caustic, resins, etc.)				None		year
Periodic replacement of blowers and other hardware	1	LS	$1,000	$1,000	per	year
Project management/reporting (for both collection system and plant)	96	hours	$110	$10,560	per	year
Disposal fees (i.e., carbon replacement):	1	LS	$20,000	$20,000	per	year
SUMMARY TOTAL O&M COSTS:				$133,560	per	year
REMEDIAL SYSTEM PERFORMANCE MONITORING COSTS						
Assume avg. 1 monitor wells per 2 SVE points, plus QC = 8 samples per quarter; quarterly sampling (32 per year)						
Analytical costs (plant performance monitoring)	32	each	$200	$6,400	per	year
Analytical costs (e.g., extraction well effluent and sewer discharge samples)				NA	per	year
Number of process monitoring points (e.g., wells/discharge points) sampled				17 per quarter		
Sampling Labor and ODCs						
Mob/demob, use 20 hrs. X $75/hr = $1500 per quarter	4	qrtr	$1,500	$6,000	per	year
Sample labor, use 8 hrs/well X $75/hr = $600	32	each	$600	$19,200	per	year
Equip. and supplies, use $500 per event	4	qrtr	$500	$2,000	per	year
Labor (data validation and evaluation)	64	hours	$75	$4,800	per	year
Project management/reporting (10% of monitoring costs)	0.1	percent	$38,400	$3,840	per	year
TOTAL MONITORING COSTS:				$42,240	per	year
MONITORING WELL NETWORK/SURFACE WATER/AIR SAMPLING COSTS (covered under groundwater costs)						
MAJOR UPGRADE COSTS (none anticipated)						

TABLE E.3
BACKUP COSTS FOR COST-TO-COMPLETE ESTIMATE (DUNN FIELD SOURCE AREA)
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

<u>FUTURE SITE CHARACTERIZATION</u>					
Detailed vadose zone investigation to define source area soils; Simulprobe with direct push sampling, followed by HAS drilling at hot spots. Cost based on engineering judgement.				$250,000	lump sum
<u>FUTURE REPORTING AND REGULATORY SUPPORT</u>					
Dunn Field Feasibility Study Report				$75,000	
Proposed Plan				$30,000	
Record of Decision				$30,000	
			SUBTOTAL:	$135,000	
Annual SVE evaluation report.				$30,000	per year
Future regulatory support (e.g., regulatory support, RAB meeting attendance, etc.) - Covered under groundwater costs.					
<u>OTHER COSTS</u>					
SVE pilot study/design	1	LS	$100,000	$100,000	

TABLE E.4
BACKUP COSTS FOR COST-TO-COMPLETE ESTIMATE (DUNN FIELD GROUNDWATER)
REMEDIAL PROCESS OPTIMIZATION
DEFENSE DEPOT MEMPHIS, TENNESSEE

CAPITAL COSTS	Quantity	Units	Unit Rate	Summary Cost	
Existing water collection/extraction/discharge system: (well installation, extraction pumps, conveyance pipelines, discharge facilities, etc.) Installed c. 1998, costs based on actual D.O.s plus 8% for U.S. Army Corps of Engineers (USACE). PAST COST, SO NOT INCLUDED IN SUMMARY PAGE	1	LS	$2,563,527	$2,563,527	
Water treatment system: (Not applicable; uses direct discharge to sewer)	0	LS	$0	$0	
Install pumps, piping and controls for 4 new extraction wells on the south end of Dunn Field. Costs based on D.O. 5 award to date, plus 8% for USACE.	1	LS	$795,356	$795,356	
Install permanent force main (based on DDMT's 5/2000 CTC).	1	LS	$495,600	$495,600	Year 2001
SUBTOTAL FUTURE CAPITAL COSTS:				$1,290,956	
TOTAL CAPITAL COSTS:				$3,854,483	

OPERATION AND MAINTENANCE (O&M) COSTS					
Based on current D.O. 1 contract costs, plus 8% for USACE. Includes performance monitoring costs. Assumes no long-term monitoring costs included.	12	month	$13,936	$167,236	per year
TOTAL O&M COSTS:				$167,236	per year

MONITORING WELL NETWORK/SURFACE WATER/AIR SAMPLING COSTS
Assume 31 wells (20 monitoring wells and 11 recovery wells) per event, plus QC = 35 samples per quarter; quarterly sampling (140 per year).
Also assumes TCE will decay to the 5 ug/L MCL by year 2039 based on first-order decay constant of 0.16 yr^{-1}.
Reduce to semiannual sampling after 5th year.

Quarterly monitoring (based on current awards costs, plus 8% for USACE)	4	qrtr	$23,015	$92,062	per year
Number of process monitoring points (e.g., wells/discharge points) sampled				31 per quarter	
Annual reporting	1	each	$26,038	$26,038	per year
TOTAL MONITORING COSTS:				$118,099	per year

MAJOR UPGRADE COSTS				Summary Cost	Estimated Completion Date
Recovery system replacement or upgrades (based on 5% of initial capital costs, applied every 5 years)	0.05	percent	$2,563,527	$128,176	Every 5 years
TOTAL MAJOR UPGRADE COSTS:				$128,176	

FUTURE SITE CHARACTERIZATION

See Dunn Field source area backup.

FUTURE REPORTING AND REGULATORY SUPPORT					
5-year reviews				$40,000	per every 5 years
Future regulatory support (e.g., regulatory support, RAB meeting attendance, etc.) (COVERED UNDER MI COSTS)	0	hours	$110	$0	per year

OTHER COSTS

None anticipated.

APPENDIX F

SUPPORTING DATA FOR ALTERNATE TCE CLEANUP GOAL CALCULATION